

DISSOLVED OXYGEN CONCENTRATION AS A POTENTIAL INDICATOR OF  
WATER QUALITY IN NEWPORT BAY:  
A REVIEW OF SCIENTIFIC RESEARCH, HISTORICAL DATA, AND CRITERIA DEVELOPMENT

Prepared for the Santa Ana Regional Water Quality Control Board

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## EXECUTIVE SUMMARY

The frequency and magnitude of macroalgal blooms in estuaries has increased worldwide as development in coastal watersheds has resulted in increased nutrient loading to these systems. One of the many ways that abundant macroalgae reduce habitat quality is through depletion of dissolved oxygen (DO) in the water column. The relationship between macroalgae and DO has led the Santa Ana Regional Water Quality Control Board (SARWQCB) to investigate the utility of using DO as an indicator for the nutrient Total Maximum Daily Load designed to protect Newport Bay from seasonal blooms of macroalgae and associated eutrophication. However, several technical issues must be resolved before a determination can be made about the viability of DO in this capacity. This document provides the first step of technical analysis by summarizing the existing literature about the causes and effects of low DO, reviewing existing DO criteria established on the east coast, and providing recommendations for additional analysis that would need to be completed as part of development of DO criteria<sup>1</sup>.

A number of factors interact to determine the amount of oxygen dissolved in surface waters. These factors include temperature, salinity, atmospheric diffusion, primary and secondary production and vertical stratification of the water column. Atmospheric diffusion and photosynthesis by primary producers are the two main processes that increase DO concentrations. Biological oxygen demand (BOD), the need for oxygen by all aerobic organisms, is one of the main processes that depletes DO in aquatic systems. Macroalgae produce oxygen via photosynthesis, yet they also respire, consuming oxygen. When algal mats are dense and lower layers are light limited, respiration can consume more oxygen than photosynthesis produces, creating a net BOD. BOD can be further increased when decaying macroalgae are colonized by bacteria, resulting in re-mineralization of nutrients to the water column. These additional nutrients can stimulate additional macroalgal production, which further increases BOD.

Vertical stratification of the water column is another important factor in the occurrence of hypoxia (functionally defined in this document as DO concentrations <3 mg/l). Vertical stratification prevents oxygen-rich water of the upper productive layer from mixing downward to replenish the lower layers where oxygen is depleted by aerobic decomposition of organic matter. When vertical stratification occurs, continuous hypoxia (>24 h) can develop. When systems are well mixed, daily turnover of the water column replenishes oxygen at depth, however cyclic hypoxia (<24 h) may still occur due to consumption of oxygen at nighttime.

Aerobic aquatic organisms need DO to survive. Insufficient DO can cause mortality of fish and invertebrates, sub-lethal effects such as reduced growth and reproduction, and changes in behavior, all of which can lead to an overall decrease in ecosystem productivity. Anoxic conditions in sediments can lead to decreases in species diversity and organism abundance, which can have profound effects on the estuarine food web. Additionally, anoxic conditions in sediments can increase benthic flux of nutrients that may further stimulate primary production, and release of compounds, such as metals and sulfides, that are toxic to aquatic organisms.

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<sup>1</sup> The term criterion is used broadly to refer to water quality indicators, objectives and US EPA water quality criteria.

Historical DO data from Newport Bay show that hypoxia may have become an issue in the 1970s when anecdotal accounts note the beginning of macroalgal bloom formation. Data collected from this period indicate that hypoxia occurred in poorly flushed areas and increased with depth in these areas. Additional data collected continuously at several stations in the 1990s indicated that hypoxia occurred 7.6% of the time and was not related to either salinity ( $r^2=0.0013$ ) or temperature ( $r^2=0.0001$ ). Hypoxia occurred more frequently at night (7 p.m. to 7 a.m. the following day) and in the summer (May through October). 1990s data also showed decreases in DO with depth, similar to 1970s data.

Comparison of Newport Bay DO data to DO data collected from several other southern California estuaries showed several similarities. The occurrence of hypoxia was strongly related to tidal flushing, DO concentrations decreased with depth, and hypoxia was more frequent at night and in the summer. In none of the systems studied in southern California has DO availability or frequency of hypoxia been quantitatively related to macroalgal abundance.

DO criteria have been developed for regions of the Atlantic coast of the US and are being used to manage water quality in the Chesapeake Bay. US EPA's suggested criteria to protect aquatic life from Maine to North Carolina for both continuous and cyclic hypoxia for juvenile and adult survival, growth effects, and larval recruitment effects are:

- If DO at a site exceeds 4.8 mg/l, then the site meets protective criteria.
- If DO is <2.3 mg/l for >24 h, then the site fails to meet objectives for protection.
- If a site has DO <2.3 mg/l for <24 h, the intensity and duration of hypoxia determine the minimum criteria.
- If the DO at a site is between 2.3 and 4.8 mg/l, the site must be further evaluated to determine the duration and intensity of hypoxia.

For the Chesapeake Bay, DO criteria were established in conjunction with water clarity and chlorophyll *a* objectives to address nutrient and sediment pollution. Criteria were developed for 5 specific ecological habitats within the Chesapeake Bay (migratory spawning and nursery, shallow-water bay grass, open-water fish and shellfish, deep-water seasonal fish and shellfish, and deep-channel seasonal refuge) that reflect the natural conditions of the system such as vertical stratification, short-term fluctuations in DO availability, and seasonal changes.

If defensible DO criteria are to be established for Newport Bay, the link between DO and the proposed cause and effect variables needs to be substantiated. The link between water quality and DO concentrations rests on the general theory that nutrient enrichment stimulates primary production, which consumes oxygen and produces a large amount of organic matter that decomposes, again consuming oxygen. To confidently establish DO criteria and use them to manage water quality and protect aquatic resources in Newport Bay, the following relationships need to be investigated:

- Correlation between nutrient loading and macroalgae abundance;
- Correlation between macroalgae abundance and DO; and
- Correlation between DO and life history needs of aquatic species

One way to establish a better relationship between DO and macroalgae would be to correlate continuous 24 hour DO data at multiple depths with quantitative measurements of macroalgal abundance.

Newport Bay differs from other systems for which DO criteria have been developed in several important ways. The first is the predominance of macroalgae and the second is differences in physical dynamics. Phytoplankton are the dominant primary producers in many East Coast estuaries whereas macroalgae are abundant and can account for a significant portion of the primary production in southern California estuaries. How the type of dominant primary producer affects the relationship between water quality and DO availability is unknown.

Vertical stratification of the water column in Newport Bay may be limited temporally and spatially. Factors limiting vertical stratification include limited formation of a salt wedge, shallow depth, and wind and tidal driven mixing that may break up any thermal stratification. Salinity induced vertical stratification may only occur during medium-sized storms and low flow conditions. Therefore, Newport Bay may often be mixed and the spatial and temporal extent of vertical stratification limited. If this is the case, then the effects of oxygen-consuming processes may not accumulate. Thus, there may not be a strong link between DO concentrations and macroalgal abundance and DO may not be the best metric for macroalgal biomass. However, if vertical stratification does occur in Newport Bay as a result of natural processes, then it should be accounted for in DO criteria.

To begin development of defensible DO criteria with which to manage water quality in Newport Bay, the following considerations need to be explored:

- The links between water quality and DO need to be substantiated if DO criteria are to be used as a metric of water quality. The relationships between nutrient loading, macroalgae and DO need to be investigated, characterized and quantified if possible.
- The effects of DO on Newport Bay species need to be established so that DO criteria protective of aquatic life can be developed.
- A better understanding of how the predominance of macroalgae (as opposed to phytoplankton) in Newport Bay makes the system different from other systems where water quality and overall ecosystem health is needed.
- The role and importance of vertical stratification in relation to hypoxia in Newport Bay needs to be understood.
- The natural variability in DO needs to be characterized.
- A monitoring plan should be devised that maximizes the chance of capturing the areas, periods, and durations when hypoxia occurs. This plan should take into account frequency of sampling, depth, and sensitive habitat.

A watershed management plan that addresses factors in the watershed that contribute to hypoxia may result in reductions in other water quality problems as well.

CHAPTER 1.  
INTRODUCTION

Nutrients, Macroalgae and Dissolved Oxygen

Eutrophication of estuaries (defined here as accumulation of large quantities of macroalgae) is an increasingly common phenomenon worldwide (McComb et al. 1981; Sfriso et al. 1987; Schramm and Nienhuis 1996; Hernández et al. 1997). Nutrient availability often regulates macroalgal abundance (Hanisak 1983; Howarth 1988) with increases in nutrient supply stimulating growth and biomass accumulation (Harlin and Thorne-Miller 1981; Valiela et al. 1992; Fong et al. 1993; Peckol et al. 1994; Marcomini et al. 1995; Hernández et al. 1997). The occurrence of macroalgal blooms in estuaries is closely related to increased nutrient loading to coastal waters (Valiela et al. 1992, 1997; Hauxwell et al. 1998; Flindt et al. 1999). Blooms are often composed of opportunistic macroalgae in the genera *Enteromorpha* and *Ulva*, which have high nutrient uptake rates (Rosenberg and Ramus 1984; Fujita 1985; Pederson 1994) and the ability to store nutrients (Hanisak 1983; Fujita 1985; Fong et al. 1994) allowing them to proliferate in temporally dynamic systems where nutrient inputs can be pulsed and episodic.

Macroalgae are not inherently problematic in estuarine ecosystems, in fact they have many positive effects. Macroalgae are an important input to estuarine food webs (CDFG 1989; Kwak and Zedler 1997) as they are a food source for snails (Soulsby et al. 1982), fish (Allen 1994) and crustaceans (Boyer 2002). Algal detritus is also an important input to the microbial food web (Tenore 1977; Hull 1987). Macroalgae provide protection from predation for several commercially important species such as lobsters (Smith and Herrnkind 1992) and crabs (Wilson et al. 1990). Mats of macroalgae can also slow current speeds, which may enhance settlement of planktonic larvae (Hull 1987).

However, macroalgal biomass often accumulates, at least seasonally, to levels where it negatively impacts estuarine ecosystems and reduces habitat quality. Increases in macroalgae have been linked to loss of seagrass habitat via shading (Valiela et al. 1997; Flindt et al. 1999) as well as profound changes in biogeochemical cycles (Valiela et al. 1992, 1997; Sfriso et al. 1987, 1992; Flindt et al. 1999; Krause-Jensen et al. 1999). Blooms are also known to deplete the water column and sediments of oxygen (Sfriso et al. 1987; Valiela et al. 1992), leading to changes in species composition and shifts in community structure (Raffaelli et al. 1991; Bolam et al. 2000).

There appears to be a significant link between anthropogenic change in watersheds, nutrient loading to coastal waters, increased primary productivity and decreases in dissolved oxygen (DO) availability. There has been a trend toward increased frequency and intensity of bottom water hypoxia in many coastal areas (Kemp and Boynton 1980; Parker and O'Reilly 1991; NOAA 1996; Janicki et al. 2001; US EPA 2003) concurrent with increased anthropogenic nutrient loadings and eutrophication in coastal waters over recent decades (Nixon 1995; Rabalais et al. 2002). The issue is paramount as noted by Diaz and Rosenberg (1995):

*“There is no other environmental variable of such ecological importance to coastal marine ecosystems that has changed so drastically in such a short period as dissolved oxygen.”*

*Recent Water Quality Management Actions in Newport Bay*

Newport Bay (Bay) is the second largest estuarine embayment in southern California and provides critical natural habitat for terrestrial and aquatic species. The lower portion of the Bay is one of the largest small craft harbors in California and residences and commercial properties line its shores. The upper portion of the Bay is a state ecological reserve and serves as refuge, foraging areas, and breeding grounds for a number of threatened and endangered species. The Bay also provides significant spawning and nursery habitats for commercial and non-commercial fish species. The Bay is connected to San Joaquin Marsh, the largest coastal freshwater marsh in southern California. The primary freshwater input to the Bay, San Diego Creek, provides a corridor for wildlife movement between the Bay, San Joaquin Marsh, and upland areas. These diverse habitats make the Bay an important ecosystem within the urban landscape of southern California.

The Bay is subject to high nutrient loads from its highly urbanized and agriculturally cultivated watershed (US EPA 1998). The combination of nutrient-rich surface water inputs, relatively warm water of the Bay, and the mild Mediterranean climate of southern California result in the excessive growth of macroalgae (Kamer et al. 2001). Prior to 1998, historical data indicated that narrative water quality objectives regarding algae and numeric objectives for nutrients in San Diego Creek were not being achieved, resulting in negative impacts to beneficial uses in the Bay. As a result, the Bay was listed on the 1998 Clean Water Act 303(d) list for nutrient impairment. The 303(d) listing precipitated the development and adoption of a Total Maximum Daily Load (TMDL) for nitrogen (N) and phosphorus (P) for the Bay in 1998, which was intended to limit nutrient inputs to the Bay (US EPA 1998). Eutrophication of the Bay continues, however, during the implementation phase of the TMDL. Until management practices in the watershed to reduce N and P loading are fully phased in, nutrient loading to the Bay may exceed the ultimate load allocation.

It is possible that macroalgal blooms in the Bay resulting from nutrient over-enrichment from the watershed adversely impact a number of the Bay's beneficial uses through reduction in water column dissolved oxygen (DO) concentrations. If so, then decreases in nutrient loading to the Bay resulting in decreased macroalgal biomass should ultimately increase oxygen availability. Thus, the availability of oxygen may be a good indicator of water quality and eutrophication in the Bay. Improvement in dissolved oxygen levels in the water column may indicate that the trophic status of the Bay is improving and moving along a trajectory toward a less-impacted state. Furthermore, DO concentrations in the Bay could be useful in assessing whether the loading capacity specified in the nutrient TMDL is appropriate to protect the beneficial uses.

There is currently no established numeric objective for DO in the Bay. The Basin Plan narrative objective states *"The dissolved oxygen content of enclosed bays and estuaries shall not be depressed to levels that adversely affect beneficial uses as a result of controllable water quality factors."* The Basin Plan numeric objectives for inland waters of 5 and 6 mg/l for WARM and COLD designated beneficial uses, respectively, do not apply to the Bay habitats, which are classified with MARINE and ESTUARINE designations. The California Ocean Plan has a quasi-numeric objective that requires knowledge of the natural DO concentration: *"The dissolved oxygen concentration shall not at any time be depressed more than 10 percent from that which occurs naturally, as the result of the discharge of oxygen demanding waste materials."*

The Santa Ana Regional Water Quality Control Board (SARWQCB) will evaluate the use of DO as an indicator of water quality for the nutrient TMDL. At least four steps would be required to develop scientifically defensible DO objectives:

- Review of available literature relating to DO,
- Collection of data in the Bay to characterize the spatial and temporal patterns of DO concentrations,
- Establishing the impact of low DO on specific beneficial uses, and
- Identification of the specific causes of low DO.

The first step of this process, a literature review, is presented here. The goal of this review is to provide assistance to the SARWQCB in developing DO as a water quality indicator for the Newport Bay Watershed Nutrient TMDL and to help focus future research efforts on the most critical questions. This review synthesizes the current state of knowledge from peer-reviewed scientific literature on known causes and effects of low DO availability in estuarine and coastal marine systems, historical data of DO concentrations in the Bay, relevant data from other southern California estuaries, and summarizes methodologies from relevant documents guiding development of DO criteria in other systems. The review concludes with recommendations of assessments and considerations necessary to guide development of DO as an appropriate indicator of nutrient enrichment and eutrophication in Newport Bay.

#### Organization of Review

This literature review is organized into six chapters:

- Chapter 1 provides an introduction to the overall problem of nutrient loading, eutrophication and DO availability in estuaries worldwide, and the context of the problem in Newport Bay
- Chapter 2 presents the mechanisms affecting availability of DO in aquatic systems.
- Chapter 3 reviews effects of hypoxia on aquatic organisms and physical components of estuarine systems.
- Chapter 4 reviews historical DO concentrations in Newport Bay and other regional estuaries.
- Chapter 5 summarizes US EPA's approaches to DO criteria development for the East Coast of the US and the Chesapeake Bay.
- Chapter 6 summarizes the findings of the literature review in the context of DO criteria as a water quality management tool and highlights remaining questions that need to be answered.

## CHAPTER 2. PROCESSES AND MECHANISMS THAT REGULATE DISSOLVED OXYGEN IN AQUATIC SYSTEMS

The first step in understanding how DO criteria can be used to manage water quality is to understand some of the basic properties of oxygen dissolved in surface waters and the factors that affect its availability. In this chapter we summarize scientific literature from peer-reviewed journals and technical reports to provide background on the various processes and mechanisms that regulate DO concentrations.

### Dissolved Oxygen in Surface Waters

The amount of oxygen dissolved in water can be measured either in concentration (mg/l or ppm) or percent saturation. Hypoxia (the state where oxygen supply to organisms is deficient) is generally defined as DO below a given concentration or saturation (Table 2.1), and anoxia is the absence of oxygen. For the purpose of this document, we consider DO concentrations <3 mg/l to be hypoxic. As saturation depends greatly on environmental variables (such as temperature), concentration is a more standard measure (Turner et al. 1987; Parker and O'Reilly 1991; Stanley and Nixon 1992) and is more easily compared across systems and between studies.

The concentration of DO in water is affected by the physical and chemical properties of the water, factors that act to increase DO concentration in water, and factors that act to decrease DO concentration in water. Two of the key factors that influence DO concentration are temperature and salinity, which influence the solubility of oxygen in water. Figure 2.1 illustrates an example of relationships between DO concentration, saturation, temperature and salinity; higher temperature and salinity result in lower DO solubility<sup>2</sup>. Additional factors such as atmospheric diffusion, primary and secondary production, and water column vertical stratification also affect DO concentration and are discussed in more detail below.

The two main factors that contribute DO to the water column are 1) diffusion across and mixing at the air-water interface, and 2) photosynthesis. At the air-water interface, gases, such as oxygen, diffuse across a concentration gradient, generally enriching surface waters with oxygen from the atmosphere. Additionally, turbulence at this interface can also incorporate gases into surface waters. Photosynthesis by primary producers, such as phytoplankton, macroalgae, microbenthic algae and submerged aquatic vegetation, also provides a significant quantity of oxygen to the water column. This oxygen production is limited to the euphotic zone. Diffusion and mixing within the water column transfer oxygen from the surface layers to deeper layers.

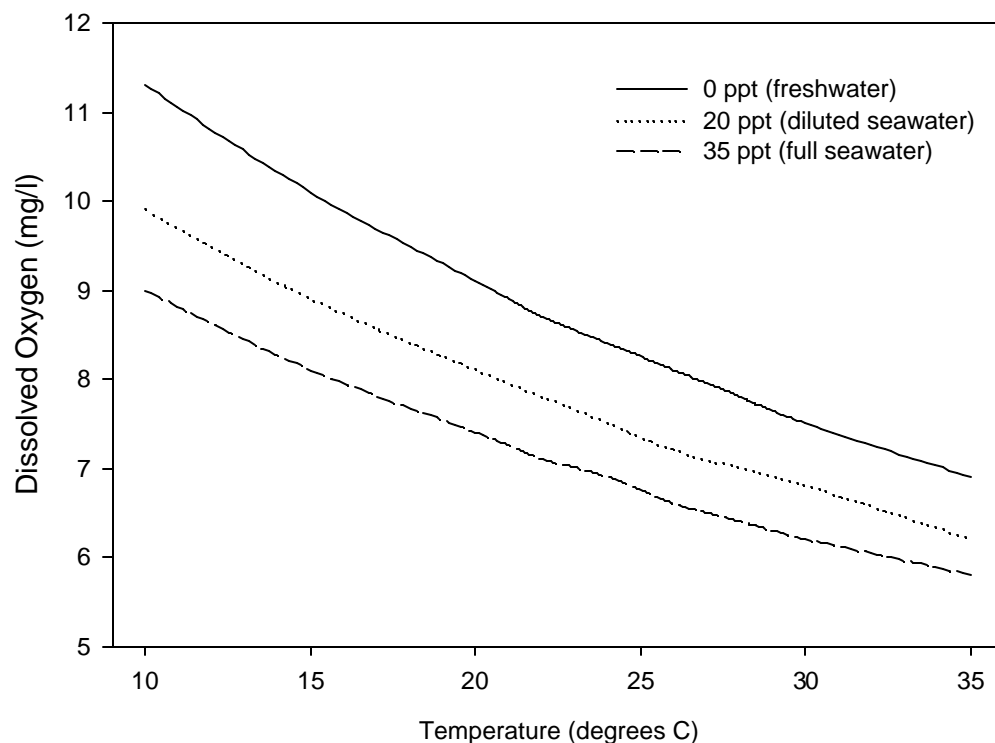
### Factors that Decrease Dissolved Oxygen

Dissolved oxygen differs from other water quality parameters in that depletion of oxygen from the water column is caused by biological oxygen-demanding processes or physical mechanisms rather than input of a substance such as a toxic chemical (US EPA 2000). Hypoxia can be a natural phenomenon in some systems, though its occurrence may presently be exacerbated by

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<sup>2</sup> See Standard Methods for the Examination of Water and Wastewater (APHA 1998) for more detailed analysis of these relationships.

human activities. The mechanisms by which oxygen is depleted from surface waters are discussed below.



**Figure 2.1. Dissolved oxygen concentrations constituting 100% saturation at sealevel (760 mm Hg) across a range of temperatures for freshwater, diluted seawater and full strength seawater.**

### Biological Processes

Aerobic organisms require sufficient oxygen for normal mitochondrial function and energy production (ATP). Thus, all aerobic, heterotrophic organisms such as bacteria, fish and invertebrates consume oxygen and generally comprise a significant portion of the biological oxygen demand (BOD) in an aquatic system. A major cause of hypoxia is bacterial decomposition of organic matter in bottom waters: organic matter sinks to the bottom and is colonized by oxygen-consuming bacteria that use the oxygen faster than it can be replaced. This can result simply from natural processes and does not necessarily indicate a water quality problem that can be regulated. However, as nutrient inputs from watersheds increase, the production of organic matter often increases and leads to a greater occurrence of hypoxic conditions.

As aerobic organisms, macroalgae also consume oxygen through respiration in addition to producing oxygen via photosynthesis when adequate light is available. Depending on light levels and the relative amounts of respiratory oxygen demand and photosynthetic oxygen production, there is either net production or consumption of oxygen, and the latter may result in hypoxic or anoxic conditions. The association of algal mats and low DO has been documented throughout the world (Table 2.2). Hypoxia often occurs at night when oxygen production ceases

but consumption continues, depleting oxygen levels (Valiela et al. 1992; D'Avanzo and Kremer 1994; Krause-Jensen et al. 1999). Hypoxia is most likely to occur just before dawn, when oxygen consumption has been occurring all night. Thick mats of macroalgae can be self-shading and during the day photosynthesis in lower layers of the mats can be limited, resulting in hypoxia or anoxia (Lavery and McComb 1991; Sfriso et al. 1992; D'Avanzo and Kremer 1994; Flindt et al. 1999). Additionally, daytime hypoxia may develop when cloud cover or calm winds persist for several days (D'Avanzo and Kremer 1994; US EPA 2003).

The extent of oxygen depletion may be affected by algal mat density. Benthic mats reduce oxygen exchange between sediments and water (Hull 1987) and also reduce currents, creating stagnant conditions under which hypoxia/anoxia may persist (Krause-Jensen et al. 1999). Lavery and McComb (1991) measured DO in banks of *Chaetomorpha linum* of different densities and found that DO was above 5 mg/l in areas with  $\leq 312$  g wet wt  $m^{-2}$  but DO dropped to  $< 2$  mg/l in areas with dense algae ( $\geq 668$  g wet wt  $m^{-2}$ ). Additionally, oxygen depletion may not be limited to the water within the mat, but may occur in water overlying the mat as well (Valiela et al. 1992) due to diffusion of oxygen from the water column down a concentration gradient to the low-oxygen environment of the mat.

Bacterial-mediated BOD may further exacerbate hypoxia/anoxia and nutrient cycling. Macroalgae decay under low oxygen conditions (Lavery and McComb 1991; Sfriso et al. 1992; Hernández et al. 1997) and are subsequently colonized by bacteria, which in turn consume oxygen and increase the BOD (Lavery and McComb 1991; Flindt et al. 1999). Microbial mineralization of dead algae in the water column releases nutrients that can then support production in healthy algae (Flindt et al. 1999). One of the best examples of this phenomenon was in the Lagoon of Venice, Italy, in 1985 (Sfriso et al. 1987). The Lagoon was subject to chronic nutrient enrichment from the watershed resulting in hypertrophic conditions. High *Ulva rigida* biomass (6-8 kg wet wt  $m^{-2}$ ) lead to several hypoxic/anoxic events where DO dropped to 2.3, 0.5 and 0 mg/l. Following these hypoxias, inorganic nitrogen and phosphorus concentrations in the water column peaked and soon after the macroalgae re-established and increased in abundance until another hypoxic event occurred. The feedback between nutrient stimulation of primary production and release of nutrients upon decomposition of primary producers can be self-perpetuating until algal production declines due to another limiting factor, such as light or temperature.

Bacterial colonization of decaying macroalgae affects the sediments as well. Microbial activity may increase sedimentation of algal matter from the water column and provide additional substrate for bacterial respiration in sediments, again increasing oxygen demand (Lavery and McComb 1991). Additionally, decomposition of algae supplies nutrients to sediments through production of fine organic material high in nitrogen content (Tenore 1977). In a seagrass bed in Florida, porewater nutrient concentrations rose as a mat of *Microcoleus lyngbyaceus* settled over the benthos (Zimmerman and Montgomery 1984). Likely, nutrient-rich compounds released from the decomposing mat fluxed into the sediments. Such compounds may be re-mineralized in the sediments and returned to the water column as inorganic nitrogen to further stimulate primary production.

### Physical Mechanisms

Physical processes as well as biological processes can contribute to hypoxia/anoxia. Vertical stratification may be the most important physical factor in establishing hypoxia. Stratification occurs when surface waters warmed by the sun overlay deeper waters that remain cool, and mixing between these two layers is reduced. This tends to occur in months when the sun's influence is greatest. Stratification can also occur due to formation of a salt wedge, when warm, fresh, less dense water from a terrestrial source overlays cold, saline, dense oceanic water, and there is little physical mixing between these two zones (US EPA 2001). In a typical phytoplankton based system, the upper layers of the water column are the site of photosynthetic oxygen production and the lower layers are the site of oxygen-consuming respiration by aerobic microbes decomposing sunken organic material. When the system is stratified, oxygen replenishment of the lower layers is decreased or non-existent and hypoxia may occur (Turner et al. 1987; Welsh and Eller 1991; Diaz and Rosenberg 1995). Again, this natural phenomenon may be worsened by increases in nutrient loading and subsequent primary production.

Vertical stratification tends to occur in systems with low tidal energy, large freshwater inputs, deep channels, or structures that impede circulation such as sills, breakwaters or dredge spoil banks (Nixon 1988). Shallow estuaries are often not vertically stratified, as wind-driven mixing tends to homogenize the water column and break up any stratification; this limits the spatial and temporal extent of hypoxia (Stanley and Nixon 1992). However, vertical stratification of shallow estuaries can occur (Williams 1997), particularly if the system is lagoonal and the mouth closes periodically, resulting in a muted tidal influence (Ward et al. 2001). For example, in Pamlico Sound, NC, with an average depth of 2.7 m, stratification and warm water temperatures were conducive to the development of hypoxia at depth (Stanley and Nixon 1992).

The relationship between vertical stratification and persistent hypoxia is seen repeatedly in a number of different systems. In Long Island Sound, temperature-driven vertical stratification led to hypoxia in bottom waters for several months (Welsh and Eller 1991). In Mobile Bay, AL, and surrounding shelf areas, there was also a strong relationship between water column oxygen levels and density stratification, with greater changes in oxygen concentrations from the surface to the bottom occurring with greater changes in density (Turner et al. 1987). Density stratification occurs in the Chesapeake Bay year round but is more intense in the summer, which is the time period during which bottom water hypoxia is usually observed (Kemp and Boynton 1980; US EPA 2003).

### *Interaction of Biological and Physical Factors to Create Temporal Patterns in Hypoxia*

Hypoxic events are generally categorized as either continuous (or persistent), lasting >24 h, or cyclic, <24 h but likely to occur daily for a period of time. Continuous hypoxia develops in bottom waters in areas where the water column is stratified and oxygenated surface water does not mix downward. Occasionally, these hypoxic bottom waters may be advected to other areas by upwelling, wind or tidal forcing (Breitburg 2002; US EPA 2003), and hypoxia may temporarily affect upper portions of the water column as well as shallow areas.

In well-mixed systems, daily turnover of the water column prevents continuous hypoxia from developing. However, cyclic hypoxia can occur at night when cellular respiration processes

consume DO. Temporary hypoxia may also occur in the upper portions of eutrophic water bodies during the day when respiration exceeds oxygen production via photosynthesis. These short-term hypoxic events may fluctuate dielly, tidally, or with weather patterns (i.e., cloud cover would increase hypoxia whereas wind-driven mixing would reduce it).

In most estuaries, hypoxia occurs during warmer months (NOAA 1998) when vertical stratification is strongest (Stanley and Nixon 1992; Diaz and Rosenberg 1995; Breitburg 2002) and primary producers are most abundant and productive (Janicki et al. 2001). Following die-offs of large algal blooms, the entire water column may become hypoxic/anoxic (Sfriso et al. 1987; Breitburg 2002). These events may be very serious as they can persist over large areas for several weeks and even the upper portions of the water column are impacted and cannot serve as a well-oxygenated refuge (Breitburg 2002). Thus, there may be no spatial or temporal relief for the organisms and mass mortality may occur.

#### Summary

Depletion of oxygen is caused by oxygen-demanding biological processes and physical mechanisms. Rather than a problem that can be regulated directly, hypoxia is a symptom of a problem, generally nutrient over-enrichment and eutrophication. Systems most prone to hypoxia experience

- High nutrient loading that stimulates high primary productivity and creates large BOD; and
- Vertical stratification, which decreases flux of oxygen-rich waters to relieve BOD.

**Table 2.1. Suggested thresholds for hypoxia.**

<i>Dissolved oxygen concentration</i>	<i>Geographic Area</i>	<i>Reference</i>
<b>&lt;2.0 mg/l</b>	Chesapeake Bay Northern Gulf of Mexico  Tampa Bay Not specified	Diaz and Rosenberg 1995 Summers and Engle 1993; Rabalais et al. 2002 Janicki et al. 2001 NOAA 1998
<b>&lt;3.0 mg/l</b>	Long Island Sound  Not specified	Parker and O'Reilly 1991; Welsh and Eller 1991 NOAA 1998
<b>&lt;4.0 mg/l</b>	Los Penasquitos Lagoon Tijuana River Estuary	Williams et al. 1999 West et al. 2002
<b>&lt;20% saturation</b>	Pamlico River Estuary, NC	Stanely and Nixon 1992
<b>DO &lt; full saturation that impairs living resources</b>	Chesapeake Bay	US EPA 2003
<b>DO &lt; 50% saturation</b>	Not specified	Breitburg 2002

**Table 2.2. Studies documenting the occurrence of low dissolved oxygen in estuaries in relation to macroalgal mat abundance.**

<i>Geographic Region</i>	<i>Macroalgal species</i>	<i>DO (mg/l)</i>	<i>Conditions</i>	<i>Reference</i>
<b>Peel Inlet, Australia</b>	<i>Chaetomorpha linum</i>	0-1.25	Algal biomass >668 g wet wt m <sup>-2</sup>	Lavery and McComb 1991
<b>Kertinge Nor, Denmark</b>	<i>Chaetomorpha linum</i>	0	4-8 cm deep in mat	Krause-Jensen et al. 1999
<b>Waquoit Bay, Massachusetts</b>	<i>Gracilaria tikvahiae</i> , <i>Cladophora vagabunda</i>	1-3	Summer, pre-dawn, stratified system	D'Avanzo and Kremer 1994
<b>Venice Lagoon, Italy</b>	<i>Ulva rigida</i>	0	Algal biomass 800 g dry wt m <sup>-2</sup>	Sfriso et al. 1992
<b>Baltic Sea</b>	<i>Pilayella littoralis</i> , <i>Ectocarpus siliculosus</i>	0.5	At night within algal mat	Aneer 1985
<b>Newport Bay, CA</b>	<i>Enteromorpha</i> <i>Enteromorpha</i> , <i>Ulva</i>	0 <3.0	Fish kill attributed to bloom Algal biomass >1.5 kg wet wt m <sup>-2</sup>	US EPA 1998 AHA 1998

### CHAPTER 3. THE EFFECTS OF LOW DISSOLVED OXYGEN ON ESTUARINE ECOSYSTEMS

After understanding the processes that reduce DO availability in estuarine systems, the next step is to understand the effects that hypoxia has on aquatic organisms and the biogeochemistry of these systems. In this chapter we summarize scientific literature from peer-reviewed journals and technical reports to provide background on the biological and chemical effects of hypoxia on estuarine systems.

#### Biological Effects of Low Dissolved Oxygen Availability

The response of aquatic organisms to low DO will depend on the intensity of hypoxia, duration of exposure, and the periodicity and frequency of exposure (Rabalais et al. 2002). Organisms have developed several physiological and behavioral adaptations to deal with temporary periods of low oxygen availability. Organisms can: 1) temporarily utilize anaerobic pathways to produce energy (ATP); 2) scavenge oxygen from hypoxic waters and increase the efficiency of oxygen transport to cells; 3) emigrate from hypoxic zones; or 4) reduce demand for oxygen by reducing activity. However, these are all short-term strategies and will not enable the animal to survive long hypoxic periods. If oxygen deficiency persists, death will ensue.

Hypoxia and anoxia lead to mortality of fish and invertebrates in estuarine systems. Studies worldwide have documented fish and invertebrate kills coincident with low DO levels. In Pamlico Sound, NC, hypoxia is known as “dead water” and is associated with fish kills (Stanley and Nixon 1992). Sfriso et al. (1987) noted mass mortality of macrofauna during an anoxic period following a macroalgal bloom in Italy. D’Avanzo and Kremer (1994) reported two anoxic events in which thousands of dying animals migrated to shallow areas of Waquoit Bay, MA. Ward et al. (2001) describe the death of striped mullet, jackknife clams and horn snails when DO was <1.0 mg/l in Los Penasquitos Lagoon, CA. Diaz and Rosenberg (1995) list 38 systems from Europe, North and South America, Asia and Australia in which the benthic community experienced significant mortality in response to hypoxia. US EPA (1998) attributes a fish kill in Newport Bay in May 1986 to anoxic conditions created by a bloom of *Enteromorpha*.

Sublethal effects of hypoxia include reduced growth and reproduction (US EPA 2000). In response to low DO, organisms may decrease their metabolic rate to avoid induction of anaerobic pathways; however, this strategy may reduce growth (Breitburg 2002). Alternatively, metabolic processes such as ventilation and circulation may increase in order to increase the delivery of oxygen to tissues, but this can reduce growth as well (Wannamaker and Rice 2000). Feeding may also decrease when inadequate levels of oxygen are available (Wannamaker and Rice 2000), which would also reduce growth and reproductive output.

Hypoxia can affect the behavior of organisms as well. Organisms avoid hypoxic areas that they would normally use for feeding, breeding and shelter (Wannamaker and Rice 2000; Breitburg 2002) leading to effective loss of habitat (US EPA 2003). As DO concentrations drop, so do fish abundance and species diversity (Breitburg 2002). Laboratory studies have substantiated the avoidance of water with DO <2 mg/l by larval estuarine fish (Breitburg 1994) and DO <1 mg/l by juvenile white mullet, croaker, pinfish, spot, and brown shrimp (Wannamaker and Rice 2000).

Mummichogs (*Fundulus heteroclitus*, the congener of the California killifish, *F. parvipinnis*) were the only species tested by Wannamaker and Rice (2000) that did not avoid DO concentrations of 1 mg/l; instead, the animals increased their aquatic surface respiration to compensate for low DO availability.

Emigration of fish from hypoxic areas can result in higher densities and increased competition for limited resources in well-oxygenated areas (Wannamaker and Rice 2000; US EPA 2003). In Mobile Bay, AL, a well-known phenomenon of shoreward movement of dense concentrations of fish and invertebrates (known as a “jubilee”) occurs repeatedly in conjunction with vertical stratification and hypoxia (Turner et al. 1987). This phenomenon has been observed in Los Penasquitos Lagoon as well; crustaceans gathered in shallow areas during periods of low DO availability (Williams 1997).

Hypoxia may increase the risk of predation by changing the behavior of the prey. Copepods generally feed on phytoplankton in surface waters at night and migrate vertically to deeper waters during the day to avoid visual predators. In the Chesapeake Bay, migration of copepods was interrupted by hypoxia and they congregated in an illuminated portion of the water column where they may have been more susceptible to predation by anchovy (US EPA 2000). Hypoxia may also cause infaunal organisms to extend themselves above the sediment surface in attempts to gain more oxygen; this may increase their vulnerability to predation (US EPA 2001).

Anoxic conditions in the sediments can have profound effects on the benthic fauna. Under mats, species diversity and the density of individuals may decrease (Norkko and Bonsdorff 1996). Other studies have found increases in abundances of opportunistic, hypoxia-tolerant species such as the polychaete *Capitella capitata* and the snail *Hydrobia ulvae* in highly reduced sediments under algal mats (Soulsby et al. 1982; Raffaelli et al. 1991) at the cost of the loss of rare species and changes in benthic community structure (Thrush 1986). These changes in the benthic infauna community may have repercussions throughout the food web, particularly for the shorebirds that depend on intertidal mudflat infauna as a primary food source (Raffaelli et al. 1989).

In summary, hypoxia in the water column can lead to mortality or sublethal impacts such as reduced growth and reproduction. Organisms may avoid hypoxic areas leading to effective habitat loss. Benthic infaunal communities may be affected as well. Ultimately, mortality of fish, invertebrates, fish eggs and larvae leads to reduced populations. Combined with reduced growth and reproduction and avoidance of hypoxic areas, the results are reduced diversity, abundance and production in hypoxic areas and an overall decrease in ecosystem productivity. Furthermore, the effects of hypoxia on resident estuarine organisms can impact the food web and trophic interactions among species (Breitburg 2002).

#### Chemical Effects of Low Dissolved Oxygen Availability

The anoxic conditions in the sediments promoted by macroalgal mats can stimulate benthic nutrient flux. Anoxic conditions promote a reducing environment stimulating a suite of chemical reactions. Ammonium (NH<sub>4</sub>) diffuses to the sediment surface from deeper layers and can then be release to the water column (Mitsch and Gosselink 1993). Phosphorus adsorbed to the surface

of sediments and complexed with iron and other elements is released (Mitsch and Gosselink 1993) and diffusion of nutrients into the overlying water column increases (Nowicki and Nixon 1985; Jorgensen 1996; US EPA 2001). Flux of inorganic N and P from sediments to the water replenishes the source of nutrients to the macroalgae, which may in turn increase growth, respiration, and BOD. Hypoxia also inhibits nitrification/denitrification (Jorgensen 1996; US EPA 2001), thereby decreasing nitrogen loss from the system and increasing ammonia recycling. These mechanisms of positive feedback between the algae and the sediments may continue until algal growth or sediment hypoxia cease for other reasons such as light or temperature limitation.

Hypoxic conditions also promote sulfate reduction resulting in production of hydrogen sulfide, which is toxic to many organisms. Additionally, the reduction of metals such as manganese and iron can increase their solubility and availability to organisms and their concentrations may reach toxic levels (Mitsch and Gosselink 1993; US EPA 2001, 2003). Through these processes, low DO indirectly negatively affects estuarine organisms.

### Summary

The response of aquatic organisms to hypoxia depends in large part on the intensity and duration of the hypoxia. Depending on the degree of hypoxia, responses in estuarine systems include:

- Fish and invertebrate mortality;
- Sublethal effects on growth and reproduction;
- Changes in behavior of organisms leading to:
  - Effective loss of habitat as organisms avoid hypoxic areas and
  - Increased competition for resources as organisms emigrate to well-oxygenated areas;
  - Increased risk of predation;
- Changes in abundance, biomass and species diversity of benthic communities;
- Increased flux of nutrients and toxic metals from sediments; and
- Hydrogen sulfide production.

Combined, these responses contribute to an overall loss in ecosystem productivity.

CHAPTER 4. HISTORICAL DISSOLVED OXYGEN CONCENTRATIONS IN NEWPORT BAY  
AND OTHER SOUTHERN CALIFORNIA ESTUARIES

The preceding chapters have summarized general causes and effects of hypoxia in estuarine systems. The next step is to understand patterns of DO in Newport Bay and how they compare to DO concentrations in other southern California estuaries. In this chapter we present data on DO concentrations in Newport Bay from several studies over the last 30 years and data from more recent studies of several other southern California estuaries.

Historical Dissolved Oxygen Concentrations in Newport Bay

Quantitative historical data on DO concentrations in Newport Bay are available from the 1970s (Orange County Human Services Agency [OCHSA]) and the 1990s (Irvine Ranch Water District [IRWD]). Prior to 1970, DO concentrations in Newport Bay were generally > 5 mg/l and were sufficient to support aquatic life year-round (Hardy 1970 as cited in CDFG 1989). There are no quantitative records of macroalgal abundance in the Bay prior to 1970, and anecdotal accounts indicate that macroalgal blooms became noticeable during the 1970s.

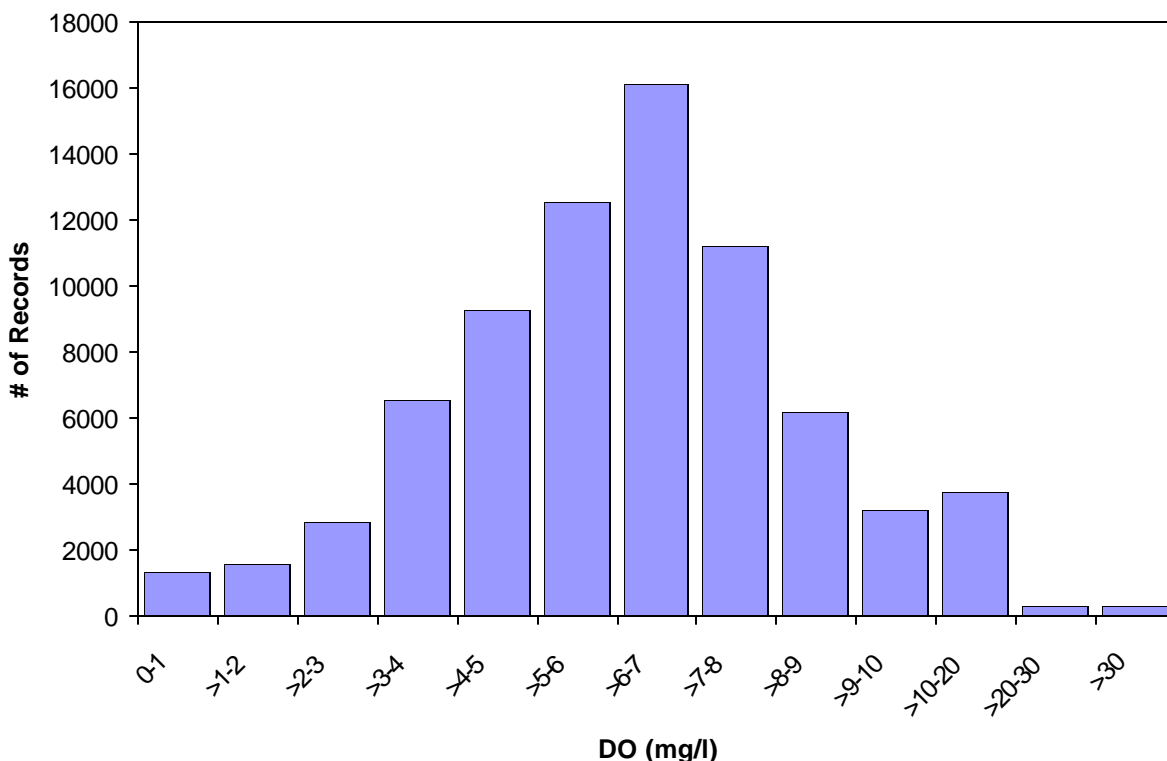
OCHSA measured DO concentrations monthly at throughout the water column at 8 stations throughout the entire Bay from July 1975 to July 1976 (OCHSA 1978). DO was generally above the 4-5 mg/l necessary to support marine life with a few exceptions. DO concentrations <3 mg/l were repeatedly detected at 3 stations in the Lower Bay: Rhine Channel, Turning Basin, and 43<sup>rd</sup> Street (Appendix A, Figure A1). Each of these stations was in a backwater area at the west end of Lido Island where water circulation may be greatly restricted; the 43<sup>rd</sup> Street and Rhine Channel stations were literally at the ends of dead-end waterways. Low DO availability at these sites was proposed as one reason for depauperate benthic communities (OCHSA 1978).

The lowest DO (0.4 mg/l) was measured in the bottom waters (0.2-0.5 m from bottom) of the Turning Basin station. The bottom of this station was in a depression, which may have further reduced flushing and water exchange with more oxygenated waters. DO also decreased with depth at several stations including Rhine Channel where monthly mean bottom water DO concentrations were around 3 mg/l in April, May and June 1976. At the 43<sup>rd</sup> Street and Turning Basin stations, DO was lower during a storm than dry weather.

Twenty years later another large data set was collected by IRWD, which began monitoring DO in Upper Newport Bay (UNB) in 1997 as part of required mitigation for proposed discharge of treated wastewater to San Diego Creek and ultimately the Bay. Probes measuring water temperature, salinity, pH, turbidity and DO were deployed one foot below the water surface at 5 stations in UNB between 1997 and 2000 (Appendix A, Figure A1). Data were recorded at 30-minute intervals and stored on dataloggers, which were downloaded every 14 days. Probes were cleaned of fouling organisms and checked for proper functioning at these times. Probes were re-calibrated as necessary. This is the only Newport Bay study we are aware of in which DO was recorded continuously and during potentially critical pre-dawn periods.

SCCWRP analyzed over 75,000 records of DO in UNB from the IRWD study. Values of DO ranged from <1 mg/l to >30 mg/l (Figure 4.1). Measurements of water supersaturated with DO

indicate a high degree of primary production<sup>3</sup>. High primary production (and subsequent decomposition of organic matter) is likely the cause of the hypoxia rather than direct input of organic matter, which would decompose but not photosynthesize and produce oxygen leading to supersaturation of the water column.

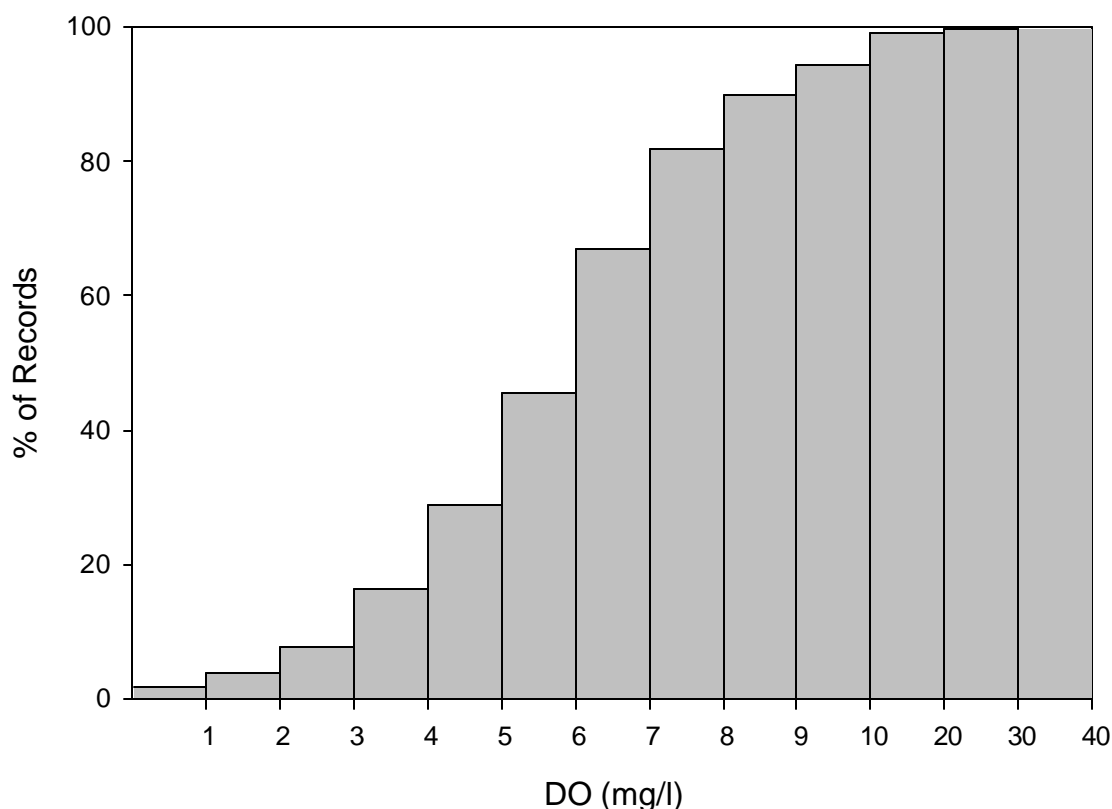


**Figure 4.1. Histogram of dissolved oxygen records from Upper Newport Bay. Data were collected by IRWD between 1997 and 2000. Values greater than 10 mg/l were grouped into intervals of 10.**

Over 92% of the DO measurements were >3 mg/l (Figure 4.2) with hypoxia (DO < 3mg/l) occurring 7.7% of the time (Table 4.1). There was no relationship between DO and either salinity ( $r^2=0.0013$ ) or temperature ( $r^2=0.0001$ ). These measurements were taken 1 foot below the surface of the water and temporal and spatial patterns of DO availability may have been different with depth. The frequency of hypoxia in Newport Bay in these data is low relative to other systems. For example, in the Mississippi River plume in the Gulf of Mexico, bottom water hypoxia occurred 50 to 80% of the time (Rabalais et al. 1994). Hypoxic conditions occurred <1% to 26% of the time in Tampa Bay, FL, in different areas of the Bay (Janicki et al. 2001). The stations where hypoxia was more frequent were deeper, exhibited a high degree of vertical stratification, had finer grained sediments, and had higher primary productivity than stations without hypoxia; there was no relationship of hypoxia to temperature or salinity (Janicki et al. 2001).

<sup>3</sup> High DO concentrations could also possibly be a result of instrument error.

Hypoxia was not evenly distributed among stations (Table 4.1). Station C experienced the greatest proportion of hypoxia and C2 experienced none, however relatively few records were available for the latter station and for station C5. Of stations C, D, and F, for which many data are available, C was closest to San Diego Creek, the primary freshwater source to the Bay. Hypoxia may have occurred more frequently at station C, due to its proximity to the loads of freshwater, sediment, and organic matter delivered to the Bay via the Creek, than at stations further down-estuary where the loading from the Creek was diluted by mixing with tidal waters. Additionally, there was less tidal flushing at station C compared to stations D and F, and the tidal prism above the salt dikes may have been reduced. These factors may have all contributed to increased frequency of hypoxia at station C.



**Figure 4.2. Cumulative frequency distribution of dissolved oxygen records from Upper Newport Bay. Values greater than 10 mg/l were grouped into intervals of 10.**

A smaller percentage of the hypoxia occurred during the daytime (7 a.m. to 7 p.m.) than the nighttime (7 p.m. to 7 a.m. the following day) (Table 4.1). The distribution of hypoxia between days and nights was similar at stations C, D and F. A different pattern was seen at station C5 but much less data were available from this station so results should be interpreted cautiously. A portion of this data was previously analyzed in another study (AHA 1998) and it was determined that hypoxia occurred during low tides at night.

**Table 4.1. Summary of records available from 5 stations in Upper Newport Bay. Data collected by IRWD between 1997 and 2000. Hypoxia defined as dissolved oxygen <3 mg/l.**

<i>Station</i>	<i># Records</i>	<i># Records &lt;3 mg/l DO</i>	<i>% Records &lt;3 mg/l DO</i>	<i>% Hypoxia Occurring</i>	
				<i>Daytime</i>	<i>Nighttime</i>
C2	1568	0	0.0	-	-
C5	1445	10	0.7	20	80
C	31049	3084	9.9	44.1	55.9
D	23937	1718	7.2	41.3	58.7
F	17331	953	5.5	42.0	58.0
<b>TOTAL</b>	<b>75330</b>	<b>5765</b>	<b>7.7</b>	<b>42.8</b>	<b>57.2</b>

To investigate the relationship between season and hypoxia, we divided the months of the year between two seasons: winter (November through April), and summer (May through October). This division is based in part on patterns of macroalgal abundance in UNB (Kamer et al. 2001). A larger percentage of the hypoxia occurred in the summer compared to winter though collection of all records was distributed evenly among these two time periods (Table 4.2). An independent concurrent study by UCLA<sup>4</sup> measured higher macroalgal biomass in June and September 1997 than in months sampled during winter (Kamer et al. 2001). Higher abundance of macroalgae could in part explain the greater occurrence of hypoxia in summer. Additionally, increased temperatures in summer would decrease the solubility of oxygen in water and could have contributed to increased frequency of hypoxia, though as noted above, temperature explained almost none of the variability in DO. Lastly, the thermal vertical stratification that often results in hypoxic conditions tends to occur more in warmer summer months; if this phenomenon is pronounced in Newport Bay in summer, it could have contributed to hypoxia as well. Similarly, the lowest bottom water DO concentrations in Tampa Bay were measured in the months of July and August (Janicki et al. 2001). Probable reasons for this pattern are increased temperatures and primary production (Janicki et al. 2001).

**Table 4.2. Seasonal distribution of hypoxia among 4 stations in Upper Newport Bay between 1997 and 2000.**

<i>Station*</i>	<i>% of All Records Collected</i>		<i>% of Records DO &lt;3 mg/l</i>	
	<i>Summer</i>	<i>Winter</i>	<i>Summer</i>	<i>Winter</i>
C5	0	100	0	100
C	53.7	46.3	69.0	31.1
D	50.1	49.9	45.4	54.6
F	50.6	49.4	55.9	44.1
<b>TOTAL</b>	<b>50.7</b>	<b>49.3</b>	<b>59.9</b>	<b>40.1</b>

\*C2 was not included as no hypoxia was recorded.

Dissolved oxygen was also measured at increasing depths at five stations in UNB over 15 of 18 sampling events from July 2000 to June 2001 (Appendix A, Figure A1) (PFRD 2001). Measurements were taken at one-meter intervals from the surface with increasing depth. DO

<sup>4</sup> The study period was December 1996 through March 1998.

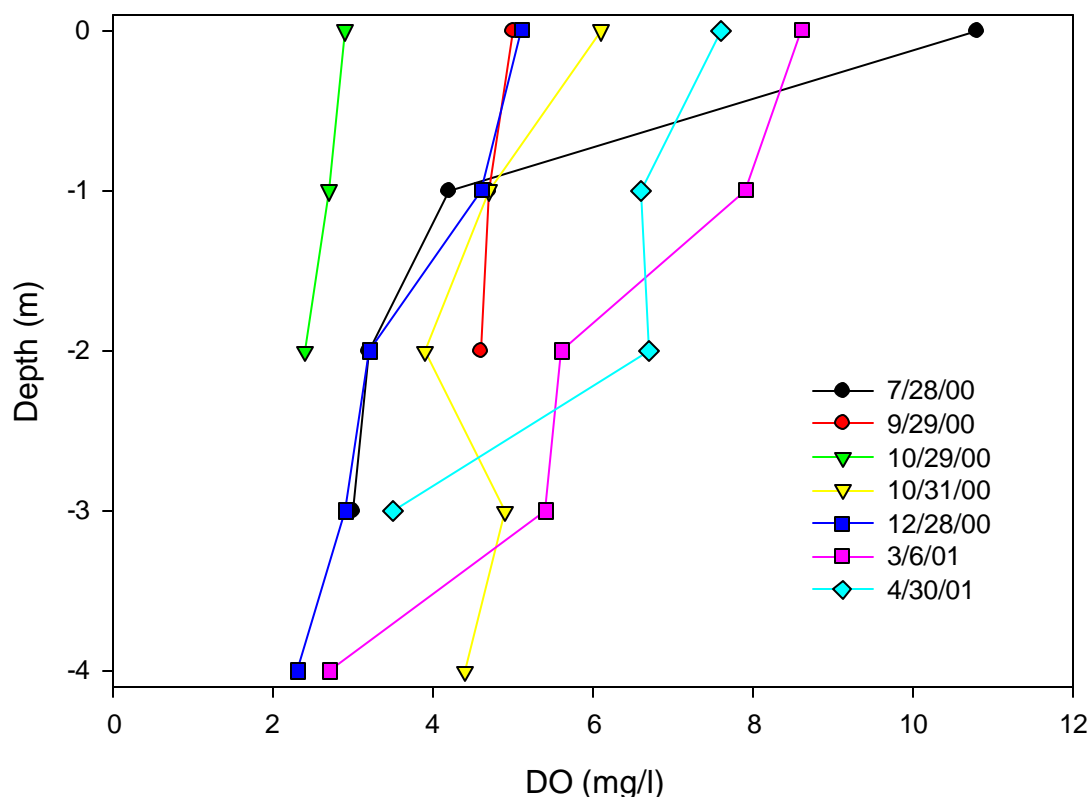
ranged from 1.4 to 28.2 mg/l and there was a trend of decreasing DO with increasing depth on a number of sampling events (Figure 4.3). Often, surface waters were well-oxygenated but measurements at 3 m depth were <3 mg/l, indicating hypoxia.

Based on the available data for Newport Bay, it appears that hypoxia became an issue in the Bay in the 1970s, the time period when anecdotal accounts of macroalgal blooms in the Bay begin.

The following general patterns were observed from the IRWD data:

- 7.65% of the continuous records collected in UNB by IRWD indicated hypoxia;
- Hypoxia was most frequent at the site closest to the head of the Bay;
- Hypoxia occurred more frequently at night than during the day;
- Hypoxia occurred more frequently in the summer months than during winter months; and
- DO decreased with increasing depth.

Based on the OCHSA study (1978), hypoxia may occur due to reduced water circulation in areas where flow is restricted. Though general patterns relating to the frequency of hypoxia detected in the IRWD study can be identified, there is no data available to attempt to identify the cause of hypoxia. Data relating DO to immediate potentially causal variable, such as primary producer abundance, is needed.



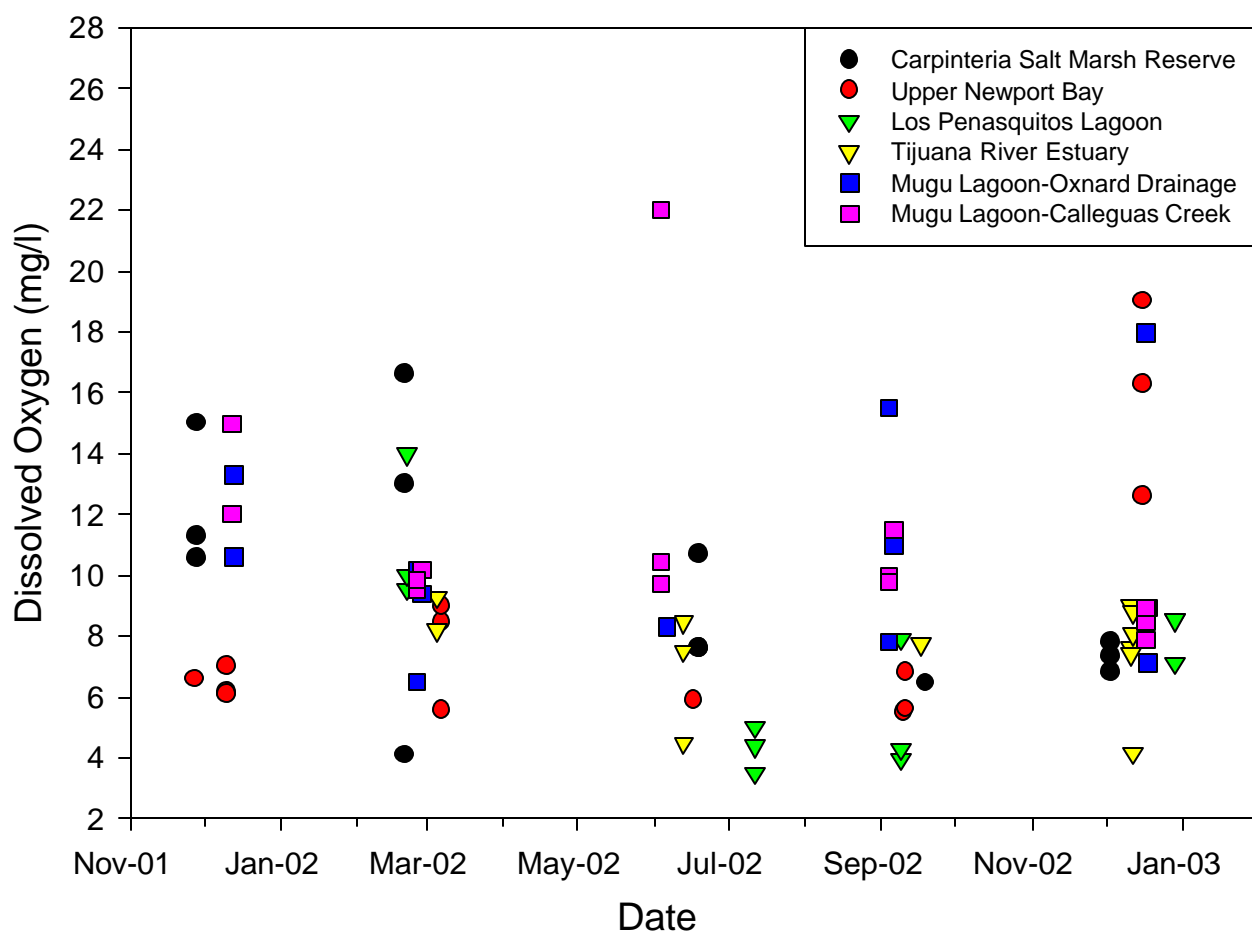
**Figure 4.3. Upper Newport Bay Unit I Basin dissolved oxygen data from selected sampling dates showing a trend in decreasing oxygen concentration with depth**

Historical data in southern California estuaries

DO data were obtained from a suite of different studies conducted in southern California estuaries. Newport Bay and the estuaries discussed below range in size, land use practices within the watershed, and nutrient regimes. However, they are all relatively shallow, tidally dominated, and experience seasonal macroalgal blooms. Generally, they are more similar to each other than any one of them is to East or Gulf Coast systems where the majority of DO research has been conducted.

Five southern California estuaries

Dissolved oxygen has been measured in 5 southern California estuaries from November 2001 through December 2002 as part of a study by UCLA and SCCWRP. The study investigates relationships between nutrient availability and response variables, including macroalgal community characteristics and DO levels. The study includes Upper Newport Bay as well as Carpinteria Salt Marsh Reserve, Mugu Lagoon (which has two main arms, Calleguas Creek and Oxnard Drainage), Los Penasquitos Lagoon, and Tijuana River Estuary (Appendix A, Figure A2).

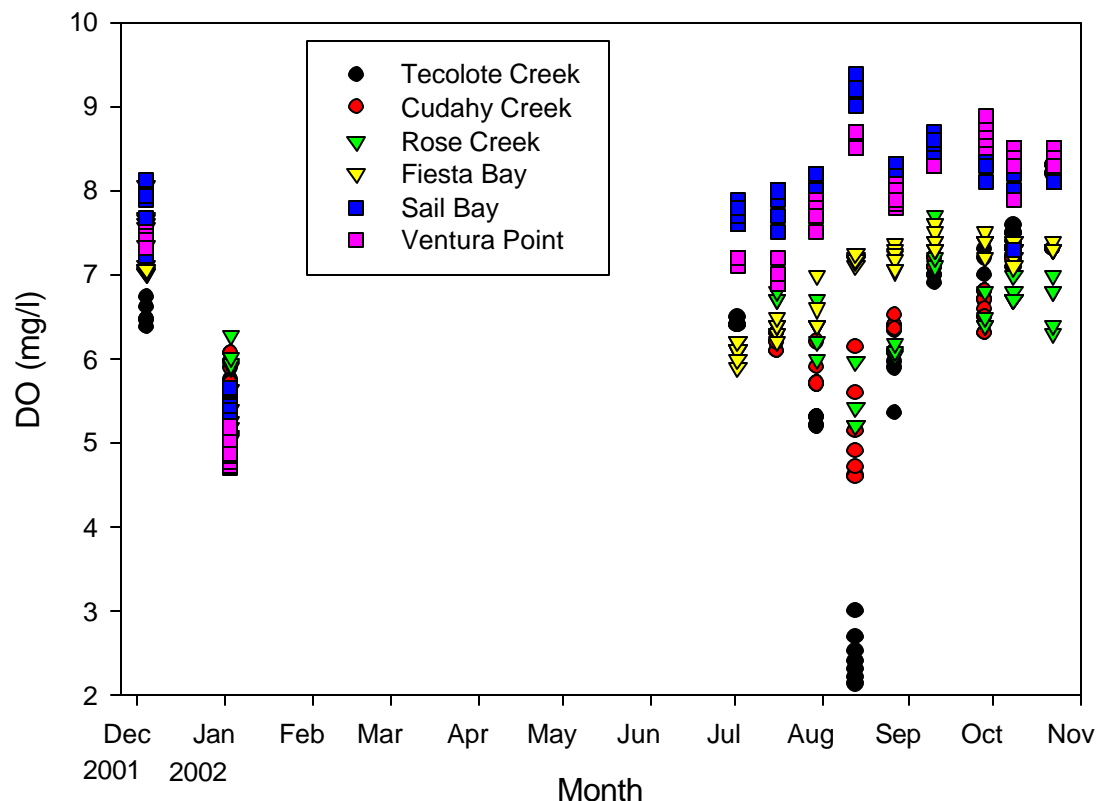


**Figure 4.4. Individual dissolved oxygen measurements from 3 transects in each of 5 southern California estuaries from November 2001 through December 2002. Measurements were taken during the daytime between 8 am and 3 pm.**

Individual DO measurements in all the estuaries were greater than 3 mg/l (Figure 4.4). There were no relationships between DO and either salinity ( $r^2=0.00296$ ), temperature ( $r^2=0.00541$ ), or macroalgal abundance (metric: % cover,  $r^2<0.001$ ). There were no seasonal differences in DO ( $p=0.231$ , Kruskal-Wallis rank test) and variability was high within each season. However, all data are from single measurements taken during the daytime in intertidal areas that drain completely daily (K. Kamer, pers. obs.). These values may not reflect hypoxic events that might occur at nighttime or in deeper waters.

### Mission Bay

Dissolved oxygen data were collected between December 2001 and October 2002<sup>5</sup> by Dr. Ronald Kaufmann of University of San Diego and analyzed by SCCWRP. DO was recorded to 3 meters depth at 0.5 meter intervals from the surface at 6 stations in Mission Bay (Appendix A, Figure A3). All data were  $>3$  mg/l with the exception of data collected from the Tecolote Creek sampling station in August 2002 (Figure 4.5). This station, at the back of Mission Bay, can have reduced circulation during periods of low tidal flushing, which may have contributed to the observed hypoxia. In several instances, DO decreased with depth though this was not common and in all cases DO was  $>3$  mg/l.



**Figure 4.5. Dissolved oxygen concentrations at 6 sampling stations in Mission Bay, CA. Data courtesy of R. Kaufmann, University of San Diego.**

<sup>5</sup> There is a gap in available data from January to July 2002

### Los Penasquitos Lagoon and Tijuana River Estuary

The Pacific Estuarine Research Laboratory (PERL) of San Diego State University has been monitoring a suite of physical, chemical and biological parameters in Los Penasquitos Lagoon (LPL) and Tijuana River Estuary (TJ) since 1987 and 1979, respectively. Annual reports dating back to the mid-1990s were obtained and the DO results are summarized below.

Over the past 9 years, LPL has experienced both persistent and diel hypoxia at multiple locations within the estuary. Persistent hypoxia tends to occur when the mouth of the lagoon closes, which happens periodically throughout the year. When the mouth closes, tidal flushing is greatly reduced, the system becomes vertically stratified, and DO concentrations, particularly at depth (5-30 cm above bottom), decline to concentrations <1.0 mg/l (Williams 1997; Williams et al. 1998, 1999; Ward et al. 2000, 2001). Mortality of striped mullet, horn snails, bubble snails, jackknife clams and other bivalve mollusks has repeatedly been associated with persistent DO declines (Williams 1997; Williams et al. 1998, 1999; Ward et al. 2001).

When the lagoon is open, hypoxia occurs on a diel cycle (Water Quality Data Datalogger Data at <http://www.perl.sdsu.edu/reports.html>). During low tides after dark in summer months, DO often drops below 3.0 mg/l and sometimes below 2.0 mg/l. Daily tidal flushing replenishes DO-depleted waters with oxygen rich oceanic waters and the hypoxia is temporarily alleviated. Additionally, biweekly data indicate that portions of the system can become vertically stratified when the mouth of the lagoon is open and that DO concentrations decrease with depth (Williams et al. 1999; Ward et al. 2000, 2001). The biweekly data do not provide information on the temporal extent of DO depressions, however we know they were episodic; 10-20 day mean DO concentrations were usually above 5.0 mg/l while minimum DO concentrations from the same periods were often <2.0 mg/l, indicating that hypoxia occurred at least once during the sampling period (Williams et al. 1999; Ward et al. 2000, 2001).

Finally, 10-21 day mean DO concentrations were generally higher in winter months than in summer months. In 2000-2001, mean DO ranged from 6.5 to 7.5 mg/l in October, December and April. From June through September, mean DO was between 3.6 and 6.3 mg/l (Ward et al. 2001). A similar pattern occurred in 2000; mean DO in March was 7.7 mg/l and from April through August it was 2.0 to 6.0 mg/l (Ward et al. 2000). In 1999, mean DO in March and April was 8.4 and 7.2, respectively, and in May, mean DO was determined twice at 4.1 and 2.7 mg/l. Biweekly data support seasonal patterns of higher DO concentrations in winter compared to summer as well (Williams et al. 1999; Ward et al. 2000, 2001).

Dissolved oxygen has been monitored continuously at TJ since 1997 and patterns similar to those at LPL have been observed. DO concentrations were generally higher in the cooler winter months than in summer months (Desmond et al. 1999, 2000; West et al. 2001, 2002). Monthly DO means were usually >3.0 mg/l however monthly minima were often <2.0 mg/l indicating that episodic hypoxic events occurred at least monthly (Desmond et al. 1999, 2000; West et al. 2001, 2002). Regular tidal flushing prevented persistent hypoxia (Desmond et al. 1999, 2000; West et al. 2001, 2002).

Some anomalous events in 1998 provide more detailed insight into DO dynamics in TJ. In October 1998, monthly mean DO at one station (the Tidal Linkage) was 1.8 mg/l. A large *Ulva*

spp. bloom preceded this low; decay of the bloom probably contributed to the observed hypoxia (Desmond et al. 1999). Examination of daily DO concentrations shows the dependence of DO on tidal flushing (Williams et al. 1998). In June 1998, persistent hypoxia developed during a neap tide series. DO was  $<2.0$  mg/l for at least 6 days straight, though the monthly mean was 3.2 mg/l (Desmond et al. 1999). In September 1998, DO concentrations  $\leq 1.0$  mg/l occurred daily for 10 days. Influx of oceanic water with incoming tides usually increased DO to  $>5.0$  mg/l and the monthly mean was 3.8 mg/l (Desmond et al. 1999). These data also demonstrate that monthly means may not adequately reflect in situ dynamics.

Measures of DO for southern California estuaries showed the following:

- The UCLA/SCCWRP study did not detect hypoxia in any of 5 southern California estuaries during 2001-2002. However, all measurements were taken during the daytime in well-flushed intertidal areas;
- Hypoxia was detected during one sampling event in Mission Bay at a poorly flushed station;
- LPL and TJ experience both persistent and diel hypoxia
  - In LPL, persistent hypoxia occurred when mouth of lagoon closed and system became vertically stratified
  - In TJ, persistent hypoxia was more rare and occurred during neap tide series when tidal flushing was reduced
  - Tidal flushing was important in relieving daily hypoxia, which occurred primarily in summer months during night-time low tides
  - Monthly DO minima data indicate hypoxia occurred episodically
  - DO availability was greater in winter than in summer

### Summary

The following similarities exist between DO patterns in Newport Bay and other southern California estuaries:

- The occurrence of hypoxia is strongly related to tidal flushing
  - In the OCHSA (1978) study of Newport Bay, hypoxia was repeatedly observed at poorly flushed stations.
  - While hypoxia was not detected in the UCLA/SCCWRP study, all measurements were taken from well-flushed intertidal areas
  - The only station in Mission Bay where hypoxia was observed was at Tecolote Creek, which is at the back of Mission Bay and has poor water circulation.
  - Continuous hypoxia occurs in LPL when the lagoon closes off from the ocean and tidal flushing ceases.
  - Reduced tidal flushing is followed by hypoxia in TJ.
- DO concentrations decreased with depth in
  - Newport Bay (PFRD 2001)
  - Mission Bay on several occasions
  - LPL (Williams et al. 1999; Ward et al. 2000, 2001)
- Hypoxia was more frequent at night in Newport Bay and cyclic hypoxia in LPL occurred at night

- Oxygen availability was greater in the winter than in the summer in LPL and TJ, and hypoxia was less frequent in the winter in Newport Bay.

In none of these systems has DO availability or frequency of hypoxia been quantitatively related to macroalgal abundance.

CHAPTER 5. US EPA'S APPROACHES AND DISSOLVED OXYGEN CRITERIA FOR THE  
EAST COAST AND CHESAPEAKE BAY

Following the discussion of the general effects of hypoxia on estuarine organisms, and with the DO regime and characteristics of Newport Bay as context, the next step is to examine established relationships between low DO availability and its effects. DO criteria have been developed for regions of the Atlantic coast of the US and are being used to manage water quality in the Chesapeake Bay. In this chapter we summarize two US EPA documents that present different methodologies for dissolved oxygen criteria development.

US EPA DO criteria for the East Coast – Source: Ambient aquatic life water quality criteria for dissolved oxygen (saltwater): Cape Cod to Cape Hatteras (US EPA 2000)

This document describes a recommended approach for deriving lower limits of DO necessary to protect coastal and estuarine fauna. This document is specifically targeted to the region of the east coast of the US from Cape Cod, MA, to Cape Hatteras, NC, but states that with appropriate modification, the recommendations may be applied to other coastal regions of the US. The approach combines aspects of traditional methodologies with a new framework factoring both DO concentration and time of exposure into the criteria. A mathematical model was used to integrate effects of low DO over time rather than deriving one number for an averaged period of time. Only species common to the Virginian Province were used in development of these criteria.

Differential DO criteria were developed for juvenile and adults life stages of organisms versus larval stages. These differential criteria are based on the theory that different life stages can withstand different degrees of mortality without significant long-term impacts to the population; therefore, the criteria developed for the most sensitive life stage may not need to be applied for the entire population at all times. For example, in nature, larval life stages suffer a high degree of mortality, and the loss of a single larva is not as significant as the loss of an individual juvenile or adult and its predicted reproductive output.

Three specific population measures for which protective criteria were developed are: 1) juvenile and adult survival, 2) growth effects, and 3) larval recruitment effects. Criteria were developed for both continuous and cyclic hypoxia. Anoxia was not considered since data on the effects of anoxia do not provide information on the threshold requirements of aerobic organisms.

The following criteria have been recommended to determine if either continuous and/or cyclic oxygen concentrations at a given site protect aquatic life<sup>6</sup>:

- If DO at a site exceeds 4.8 mg/l, then the site meets protective criteria.
- If DO is <2.3 mg/l for >24 h, then the site fails to meet objectives for protection.
- If a site has DO <2.3 mg/l for <24 h, the intensity and duration of hypoxia determine the minimum criteria.

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<sup>6</sup> Hypoxia is different from chemical constituents in that greatest response is associated with lowest concentration, not the highest.

- If the DO at a site is between 2.3 and 4.8 mg/l, the site must be further evaluated to determine the duration and intensity of hypoxia.

These criteria apply to specifically to warm saltwater but are considered protective for colder times of year as well. The derivations of these criteria are described in detail below.

### Continuous criteria

#### **Juvenile and adult survival**

The lower DO limit necessary to ensure juvenile and adult survival with only 5% mortality is 2.3 mg/l for continuous conditions. This limit is analogous to the criterion maximum concentration (CMC) for toxic chemicals except that the limit is expressed as a *minimum* since the greatest response to DO occurs when DO is lowest. This limit was derived from determination of lethal concentrations of oxygen for 12 invertebrate and 11 fish species. Below 2.3 mg/l for >24 h, juvenile and adult fish and invertebrates may suffer unacceptable mortality (>5%).

#### **Growth effects**

The DO threshold above which exposure >24 h should not negatively impact growth is 4.8 mg/l. This limit is analogous to the criterion continuous concentration (CCC) for toxic chemicals and it should protect growth and survival of most aquatic fauna in the region. Impact on growth is a sub-lethal effect of hypoxia and is generally a more sensitive test than mortality. Alternatively, impact on reproduction could be used to determine sub-lethal impacts of hypoxia; however, there are few tests of the effects of hypoxia on reproduction<sup>7</sup>. Data from 4 fish and 7 invertebrate species show that below 4.8 mg/l, growth may be reduced by 25% or more.

#### **Larval recruitment effects**

Dissolved oxygen can be less than the CCC as long as the exposure duration does not exceed a corresponding allowable number of days that ensure adequate larval recruitment. A model using 9 genera determined the effects of intensity and duration of hypoxia on larval recruitment and provides a DO limit based on the number of days a continuous exposure can occur. The cumulative fraction of allowable days above a given daily mean DO must not exceed 1.0:

$$\sum \frac{t_i(\text{actual})}{t_i(\text{allowed})} < 1.0 \quad \text{and} \quad DO_i = \frac{13.0}{(2.80 + 1.84e^{-0.10ti})}$$

where:

DO<sub>i</sub>=allowable concentration (mg/l)

t<sub>i</sub>=exposure interval duration (days)

i=exposure interval

The cumulative effects of exposure to DO below the CCC are determined by totaling the fractions of exposure duration (in days) divided by the allowable exposure duration for each DO concentration below the CCC. This incorporates duration of exposure into the DO criterion. The acceptable reduction in seasonal recruitment was 5%<sup>8</sup> and should not significantly affect the population relative to other mortality sources in the absence of hypoxia.

<sup>7</sup> The several studies that do exist show that growth is more sensitive than reproduction, therefore DO concentrations that protect growth may also protect reproduction.

<sup>8</sup> Larvae are more acutely sensitive to hypoxia than juveniles but many will suffer mortality due to factors other than hypoxia.

In summary, if DO is < 2.3 mg/l, juvenile and adult fish and invertebrate mortality may exceed 5% and protective goals are not being met. If DO exceeds 4.8 mg/l, growth is not likely to be impacted by hypoxia and most aquatic life and its uses are protected. When DO concentrations are between these values, the effect of hypoxia on larval recruitment depends on its intensity and duration (Figure 5.1).

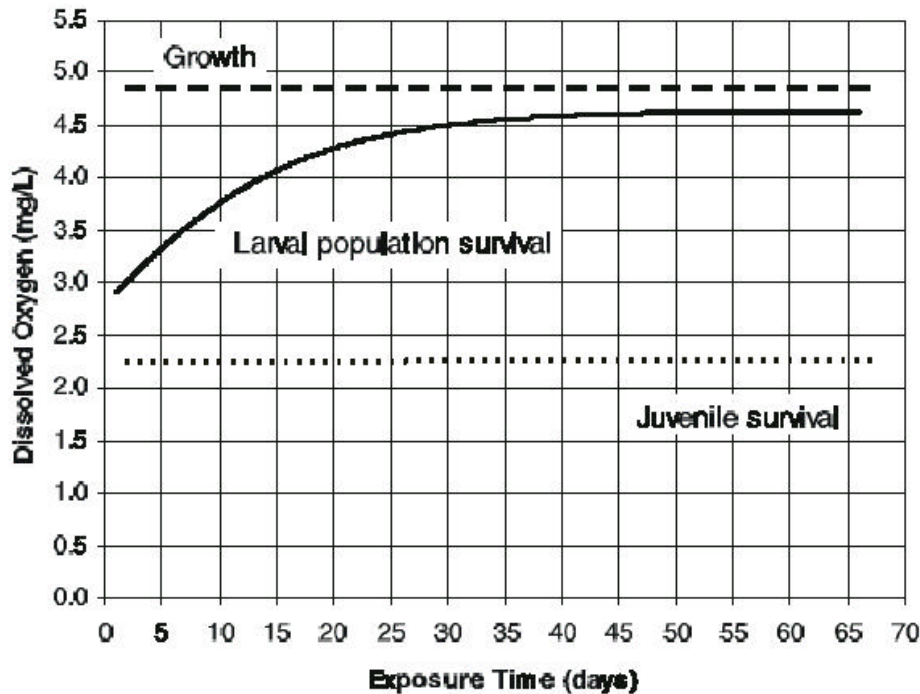


Figure 5.1. Plot of the final criteria for saltwater animals continuously exposed to low dissolved oxygen. Figure 7 in US EPA (2000).

### Cyclic criteria

#### **Juvenile and adult survival**

The criteria for juvenile and adult survival exposed to hypoxia for durations <24 h is a limit based on the hourly duration of exposure. The concentration of DO necessary for juvenile and adult survival decreases as the time of exposure to that concentration decreases. For a 24 h period, the minimum DO concentration is 2.3 mg/l, the same concentration as the continuous criteria. If the period of exposure is < 24 h, the lower DO limit is calculated as:

$$DO = 0.370 \cdot \ln(t) + 1.095$$

where DO=minimum concentration and t=exposure duration in hours.

#### **Growth effects**

The DO threshold above which exposure <24 h should not negatively impact growth varies with the intensity and hourly duration of exposure. Exposure to hypoxia for 12 of 24 h reduced growth by more than 50% in 2 invertebrates and a fish species, which implies that recovery from the hypoxia was not instantaneous once DO concentrations were adequate. The effects of

hypoxia on the animals' growth persisted even after the hypoxia was relieved; therefore the animals suffered from the effects of hypoxia more than 50% of the time even though hypoxia only occurred 50% of the time. The factor for this recovery period is 1.56 and must be included in the calculation of the reduction in growth from the intensity and hourly duration of exposure:

$$\sum_{i=1}^n \frac{t_i * 1.56 * Gred_i}{24} < 25\% \quad \text{and} \quad Gred_i = -23.1 * DO_i + 138.1$$

where:

$Gred_i$ =growth reduction

$DO_i$ =allowable concentration (mg/l)

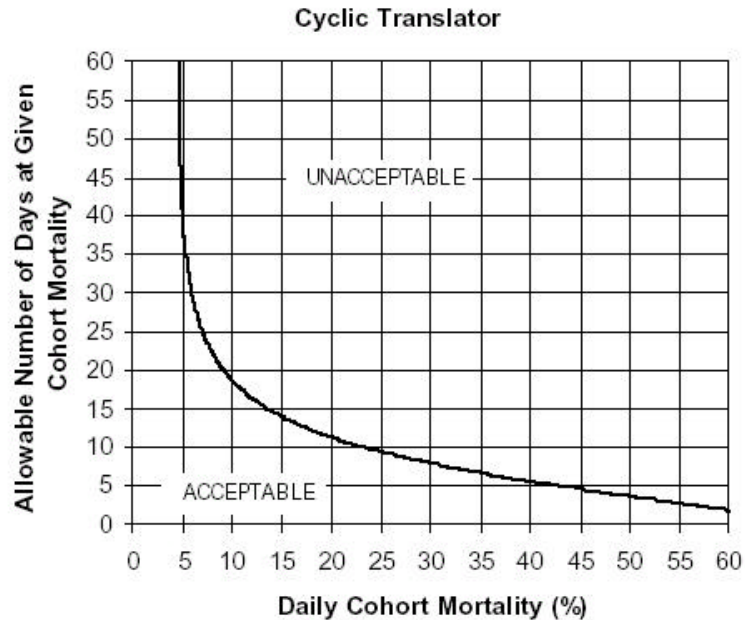
$t_i$ =exposure interval duration (hours)

$i$ =exposure interval

Acceptable reduction of daily growth is 25%; if the average daily growth is reduced by > 25%, protective goals are not being met.

### Larval recruitment effects

This criterion provides protection equivalent to the larval recruitment effects criterion for continuous exposure with the difference of recurring patterns of low DO. The minimum DO limit is based on daily cohort mortality and the allowable number of days at a given maximum daily larval cohort mortality that protects against >5% cumulative impairment of recruitment over a recruitment season. For a given DO concentration below the CCC, a certain percentage of the larval cohort will die depending on the time spent at that DO concentration. The number of days a cohort can experience daily mortality >5% without significantly impacting recruitment is determined in a cyclic translator (Figure 5.2), and this number cannot be exceeded without significant negative impact to the population.



**Figure 5.2.** Plot that combines information from a Final Survival Curve and a Final Recruitment Curve for a larval crustacean into a single cyclic translator to convert expected daily mortality from cyclic exposures into allowable number of days of those cycles. Figure 16 in US EPA (2000).

Table 5.1 summarizes the continuous and cyclic criteria for juvenile and adult survival, growth effects and larval recruitment effects.

**Table 5.1. Summary of Virginian Province saltwater dissolved oxygen criteria (US EPA 2000).**

<i>Protected Use</i>	<i>Continuous Exposure to Hypoxia (&gt;24 h)</i>	<i>Cyclic Exposure to Hypoxia (&lt;24 h)</i>
<b>Juvenile and Adult Survival</b> <i>Minimum allowable conditions</i>	Limit for continuous exposure:  DO=2.3 mg/l	Limit based on the hourly duration of exposure:  DO=0.370*ln(t)+1.095  where: DO=minimum concentration t=exposure duration in hours
<i>Growth Effects</i> <b>Maximum conditions required</b>	Limit for continuous exposure:  DO=4.8 mg/l	Limit based on intensity and hourly duration of exposure:  $\sum_1^n \frac{t_i * 1.56 * Gred_i}{24} < 25\%$ and  $Gred_i = -23.1 * DO_i + 138.1$ where: Gred <sub>i</sub> =growth reduction DO <sub>i</sub> =allowable concentration (mg/l) t <sub>i</sub> =exposure interval duration (hours) i=exposure interval
<b>Larval Recruitment Effects</b> <i>Specific allowable conditions</i>	Limit based on the number of days a continuous exposure can occur:  $\sum \frac{t_i(actual)}{t_i(allowed)} < 1.0$ and $DO_i = \frac{13.0}{(2.80 + 1.84e^{-0.10ti})}$  where: DO <sub>i</sub> =allowable concentration (mg/l) t <sub>i</sub> =exposure interval duration (days) i=exposure interval	Limit based on the number of days an intensity and hourly duration pattern of exposure can occur:  Maximum daily cohort mortality for any hourly duration interval of low DO must not exceed a corresponding number of allowable days  Allowable number of days is a function of maximum daily cohort mortality, which is a function of the DO minimum of an interval and the duration of the interval

*Chesapeake Bay Dissolved Oxygen Criteria* – Sources: Ambient water quality criteria for dissolved oxygen, water clarity and chlorophyll *a* for the Chesapeake Bay and its tidal tributaries (US EPA 2003) and [www.chesapeakebay.net/info/wqcriteria/pv/oxygen.cfm](http://www.chesapeakebay.net/info/wqcriteria/pv/oxygen.cfm)

The Chesapeake Bay is the largest estuary in the US; it is almost 200 miles long and 35 miles across at its widest point. The maximum tidal amplitude at the mouth is ~1 meter and the average depth is 6.5 meters. It is bordered by Maryland, Virginia, Delaware and the District of Columbia. The watershed of the Chesapeake Bay encompasses approximately 64,000 square miles in Maryland, Virginia, Delaware, the District of Columbia, New York, Pennsylvania and West Virginia. These states and the federal government determine the water quality standards for the Bay. DO is of concern as it is essential to the health of aquatic organisms and the frequency and intensity of hypoxic events has been increasing over the last century.

Dissolved oxygen criteria were developed in conjunction with water clarity and chlorophyll *a* objectives to address nutrient and sediment pollution. High chlorophyll *a* concentrations can reduce both water clarity and DO concentrations. Water clarity, which is essential for sufficient light to reach submerged seagrass beds, a critical habitat in the Bay, is also affected by turbidity resulting from sediment loading. Together, these three water quality criteria provide the best estimate of the impact of nutrient and sediment over-enrichment on the Bay's aquatic resources.

Dissolved oxygen criteria have been developed for 5 specific ecological uses of the Chesapeake Bay (Table 5.2). Natural processes such as high productivity in shallow waters, stratification, long residence times, low tidal energy, and nutrient regeneration may reduce DO, particularly during warmer months. Also, areas supporting fish spawning may require greater DO concentrations to achieve designated uses during winter and spring. Thus, EPA has developed criteria that protect specific aquatic habitat communities yet also reflect the natural conditions of the Bay. These criteria are designed to prevent sub-lethal effects, such as reduced growth and reproduction, and to ensure the survival of all organisms.

The criteria for each designated use were determined by the requirements of the most sensitive organisms known to inhabit each area. The criteria may therefore be over-protective for some species in a given area. For example, the minimum DO requirements for larvae in the migratory fish spawning and nursery areas range from 2.7 to 4.6 mg/l depending on time of exposure. However, an instantaneous minimum of 5 mg/l is necessary to protect early life-stage, warm-water, and freshwater species. Setting criteria using requirements of the most sensitive organisms is the most conservative way to achieve the goal of protecting all the aquatic organisms in each designated habitat.

The instantaneous minima and daily mean criteria reflect natural, short-term oxygen availability fluctuations, and seasonal criteria in deep water and channel habitats take into account the natural vertical stratification of the system. These criteria were derived in part from *Ambient aquatic life water quality criteria for dissolved oxygen (saltwater): Cape Cod to Cape Hatteras* (US EPA 2000), the approaches of which had to be modified to account for the natural variations in DO concentrations with depth and season in the Chesapeake Bay. The Virginian Province criteria were also modified with respect to the species specific to Chesapeake Bay.

**Table 5.2. Dissolved oxygen criteria for specific designated uses in the Chesapeake Bay.**

<i>Designated Use</i>	<i>Criteria</i>	<i>Qualifier</i>	<i>Rationale</i>
<b>Migratory spawning and nursery</b>	<ul style="list-style-type: none"> <li>• 7-day mean <math>\geq 6</math> mg/l</li> <li>• Instantaneous <math>\geq 5</math> mg/l</li> <li>• Shallow-water/open-water use criteria</li> </ul>	<ul style="list-style-type: none"> <li>• February-May</li> <li>• February-May</li> <li>• June-January</li> </ul>	Protect larval and juvenile freshwater species; shortnose sturgeon
<b>Shallow-water bay grass</b>	<ul style="list-style-type: none"> <li>• Open-water use criteria</li> </ul>		Protect habitat and fish and invertebrates
<b>Open-water fish and shellfish</b>	<ul style="list-style-type: none"> <li>• 30-day mean <math>\geq 5</math> mg/l</li> <li>• 30-day mean <math>\geq 5.5</math> mg/l</li> <li>• 7-day mean <math>\geq 4</math> mg/l</li> <li>• Instantaneous <math>\geq 3.2</math> mg/l</li> </ul>	<ul style="list-style-type: none"> <li>• <math>&gt;0.5</math> ppt</li> <li>• 0-0.5 ppt</li> </ul>	Ensure survival of larval and juvenile fish and invertebrates; Atlantic and shortnose sturgeon
<b>Deep water seasonal fish and shellfish</b>	<ul style="list-style-type: none"> <li>• 30-day mean <math>\geq 3</math> mg/l</li> <li>• 1-day mean <math>\geq 2.3</math> mg/l</li> <li>• Instantaneous <math>\geq 1.7</math> mg/l</li> <li>• Open-water use criteria</li> </ul>	<ul style="list-style-type: none"> <li>• June-September</li> <li>• June-September</li> <li>• June-September</li> <li>• October-May</li> </ul>	Protect eggs and larvae of bay anchovy, crabs, oysters, spot, and flounder
<b>Deep-channel seasonal refuge</b>	<ul style="list-style-type: none"> <li>• Instantaneous <math>\geq 1</math> mg/l</li> <li>• Open-water use criteria</li> </ul>	<ul style="list-style-type: none"> <li>• June-September</li> <li>• October-May</li> </ul>	Protect hypoxia-tolerant worms and clams in summer, blue crabs and finfish in winter

The Chesapeake Bay has many tidal-fresh (0-0.5 ppt) and oligohaline (0.5-5 ppt) habitats for which the above criteria are not appropriate, as they were developed for habitats with  $>5$  ppt salinity. For habitats with  $<5$  ppt salinity, the EPA freshwater dissolved oxygen criteria (US EPA 1986) are applied (Table 5.3).

**Table 5.3. Freshwater dissolved oxygen criteria (US EPA 1986).**

<i>Duration</i>	<i>Dissolved Oxygen Concentration (mg/l)</i>	
	<i>Early Life Stages<sup>1</sup></i>	<i>Other Life Stages</i>
<b>30-day mean</b>	Not applicable	5.5
<b>7-day mean</b>	6	4
<b>Instantaneous minimum</b>	5	3

<sup>1</sup>Includes all embryonic and larval stages and juvenile forms to 30 days following hatching.

Summary

US EPA's suggested DO criteria for the Virginian province (Maine to North Carolina) were:

- Developed using species common to the region but may be applied to other coastal regions of the US with appropriate modification;
- Developed for warm saltwater and are considered adequate for colder periods;
- Designed to protect various life stages (larval, juvenile, and adult) and processes (growth and recruitment) from both continuous and cyclic hypoxia; and
- Focused on cyclic and continuous hypoxia in order to incorporate both degree of hypoxia and duration of exposure into criteria.

US EPA's suggested DO criteria for the Chesapeake Bay:

- May be applied to other estuaries with appropriate modification;
- Were developed in conjunction with water clarity and chlorophyll a standards; the three objectives should be used together to determine impacts of nutrient and sediment over-enrichment;
- Were developed to protect 5 specific habitats within the system, and therefore the species in those habitats.
  - This approach takes life stage (larval, juvenile, and adult) and processes (growth and recruitment) into account to some degree, as knowledge of the use of different habitats by different life stages was included.
  - Natural seasonal variations in DO were also included in criteria.
- US EPA's freshwater DO criteria are used for areas of Chesapeake Bay with <5 ppt.

CHAPTER 6. DISSOLVED OXYGEN CONCENTRATION AS A WATER QUALITY  
MANAGEMENT TOOL IN NEWPORT BAY

There are several key considerations in developing defensible DO criteria for use in Newport Bay. If DO criteria are to be developed, a defensible link needs to be made between DO and the proposed cause and effect variables, and appropriate protocols for measuring DO need to be established. In this chapter, we discuss these considerations, as well as the ways in which Newport Bay is different from other model systems.

Substantiating Links Between Processes

There have been efforts to use DO as an indicator of nutrient over-enrichment based on the relationship between nutrient loading, primary production, and oxygen consumption (US EPA 2000, 2003). The general theory is that nutrient enrichment stimulates primary production, which consumes oxygen and produces a large amount of organic matter that decomposes, again consuming oxygen (Welsh and Eller 1991; Stanley and Nixon 1992; Breitburg 2002). Additional accompanying processes can re-mobilize nutrients creating positive feedback loops (Figure 6.1) and the cycle may continue until halted by another mechanism such as light or temperature limitation of macroalgal production.

To confidently establish DO criteria and use them to manage water quality and protect aquatic resources in a system such as Newport Bay, the following relationships need to be investigated:

- Correlation between nutrient loading and macroalgae abundance;
- Correlation between macroalgae abundance and DO; and
- Correlation between DO and life history needs of aquatic species

It is generally believed that increased nutrient loading leads to increased macroalgal abundance in southern California estuaries (UCLA/SCCWP study, unpub. data); however, this relationship has yet to be quantified. There is currently no quantitative data on the effect of macroalgae on DO concentrations in southern California estuaries or Newport Bay in particular.

UCLA/SCCWRP collected synoptic DO and macroalgal biomass data; however, because the DO data were collected during the daytime from well-flushed intertidal areas, they are not appropriate for attempting to correlate with macroalgal data. One way to establish a better relationship between DO and macroalgae would be to correlate continuous 24 hour DO data at multiple depths with quantitative measurements of macroalgal abundance.

We are not currently aware of any existing data on DO requirements or responses to hypoxia of Newport Bay or southern California species. However, data on the response of organisms from the Chesapeake Bay to low DO are available for species in the same genera as those found in Upper Newport Bay (CDFG 1989) (Table 6.1 and 6.2). It is unknown how similarly different species in the same genera might respond to hypoxia or what their DO requirements may be. Several of the species are the same in Newport and Chesapeake Bays (*Mytilus edulis*, *Capitella capitata*) and several other species (*Fundulus parvipinnis* and *F. heteroclitus*) have very similar life histories and ecological niches.

General, non-species specific criteria reviewed by US EPA (2003) are presented in Table 6.3. Adaptation of these criteria to Newport Bay would need to be preceded by confirmation that these criteria are protective of Newport Bay species. Additionally, the methodology outlined in Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses (Stephan et al. 1985) should be followed for selecting representative species in Newport Bay. Furthermore, DO criteria would need to be developed for specific ranges of temperature, pH, salinity, etc. as the sensitivity of target organisms to low DO will vary with changes in ambient conditions (US EPA 2000). For example, stressful temperature or salinity levels may influence an organism's sensitivity to low DO concentrations (US EPA 2003), and greater DO availability may be necessary to ensure an organism's survival under these conditions.

*Considerations for Adapting Existing Criteria for Use in Newport Bay*

There are two key considerations in adapting DO criteria developed elsewhere for use in Newport Bay. The first is the predominant type of primary producers, and the second is differences in the physical dynamics of southern California estuaries versus East Coast estuaries, such as the Chesapeake Bay.

Phytoplankton are the dominant primary producers in the Chesapeake Bay and in many other systems where DO concentrations have been studied (Cloern 2001) and macroalgae are minimally, if at all, important in overall ecosystem primary production. The situation is quite different in southern California estuaries, including Newport Bay. Macroalgae are abundant and can account for a significant portion of the primary production (Zedler 1980). The relationships between nutrient loading, primary production, and DO availability have for the most part been developed in phytoplankton-based systems and it is unknown how the relationships may differ when macroalgae are a dominant primary producer.

West coast estuaries also differ from those on the east coast in terms of the physical structure of the water column. In river-dominated estuaries such as the Chesapeake and many other East and Gulf Coast systems with significant year round freshwater flow, the interface between the warm, fresh water and the cold, denser seawater (the salt wedge) is a major contributing factor to vertical stratification and may extend throughout the length of the estuary. Additionally, in the Chesapeake Bay, which is considerably deeper than Newport Bay, thermal vertical stratification occurs when the sun warms the surface layers but the deeper layers remain cool.

In Newport Bay, vertical stratification of the water column may be limited temporally and spatially. Factors limiting vertical stratification include limited formation of a salt wedge, shallow depth, and mixing. In water quality model development for Newport Bay, it was determined that vertical stratification due to salinity was about 5 times more important than stratification due to temperature (RMA 2001). In Newport Bay and other southern California systems where freshwater inputs in the dry season may be minimal, the zone of the freshwater-seawater interface may be small relative to the size of the estuary. For example, in Newport Bay, this zone is mostly limited to the Unit 1 Basin, which is where decreases in DO with depth were observed (PFRD 2001). RMA determined that vertical stratification of the Upper Bay due to salinity occurred during medium-sized storms (1000-2500 cfs) and that weak vertical

stratification occurred during low flow (~15 cfs) conditions (RMA 2001). Therefore, limited salt wedge formation limits the spatial extent of vertical stratification, and areas of the estuary without the salt wedge may be consistently well mixed.

Similarly, the effects of wind and tidal driven mixing on vertical stratification of the water column in Newport Bay should be investigated. If the system is continually well mixed, thermal-driven stratification may not set up and hypoxia might not occur; however, nutrient over-enrichment and eutrophication could prevent other beneficial uses from being met. The bathymetry and tidal regime of Newport Bay combine to create very different hydrodynamics from those of the Chesapeake Bay. With a natural average depth of about 1-2 meters (with the exception of the navigational channel, which is dredged to approximately 4 meters below sea level), Newport Bay is shallow compared to the Chesapeake Bay, which has an average depth of 6.5 meters (NOAA 1998) and the mainstem is over 30 meters deep in some areas. Tidal amplitude at the mouth of Newport Bay can exceed 2 meters whereas at the mouth of the Chesapeake the maximum amplitude is just over 1 meter. In Newport Bay, the combination of a shallow system and the degree of tidal amplitude create a tidally dominated system with a large tidal prism that may often be well-mixed, thereby preventing persistent vertical stratification. In the Chesapeake Bay, smaller tidal amplitude in a much deeper system results in proportionally less of the system being exchanged with each tidal cycle, and vertical stratification occurs regularly. The different hydrodynamics of these systems may have radical impacts on the DO regimes within each, and the systems may not respond similarly to the factors that influence DO availability. If this is the case, there may not be a strong link between DO concentrations and macroalgal abundance in well-mixed systems such as Newport Bay, and DO may not be the best metric for macroalgal biomass. Similarly, a study of Tampa Bay found no relationship between external nitrogen loads and the areal extent of hypoxia (Janicki et al. 2001).

Where decreases in DO concentrations with depth were seen, such as in the Unit I Basin, it is unclear whether the DO profiles were due simply to natural vertical stratification processes or if bottom-water hypoxia was exacerbated by anthropogenic nutrient over-enrichment from the watershed. The effect of natural, seasonal vertical stratification on DO was taken into account in development of Chesapeake Bay DO criteria (US EPA 2003). There are allowances in the Basin Plan and California Ocean Plan for natural depressions of DO in surface waters. Thus, if there is a natural process that results in vertical stratification of the water column in Newport Bay, then its temporal dynamics and impact on DO concentrations must be assessed so that it can be accounted for in DO criteria. However, the causes and extent of vertical stratification in Newport Bay need to be fully understood to develop defensible DO criteria.

#### *Temporal and Spatial Considerations in Dissolved Oxygen Monitoring*

Based on RMA's determination of the occurrence of vertical stratification during medium-sized storms and low flow conditions (RMA 2001), intense study of bottom-water DO during these periods may be warranted. Whether or not the salinity induced vertical stratification leads to hypoxia needs to be determined. The relationship may depend on the duration of the stratification, tidal cycle, and season (i.e., primary producer abundance).

In addition to establishing links between nutrients, DO, and macroalgae, the way in which DO data are collected to determine if criteria are being met may influence the conclusion drawn from the data. For example, the frequency with which DO is measured can have profound effects on the conclusions drawn and the subsequent characterization of a system as either healthy or impaired. Summers and Engle (1993) evaluated different sampling strategies used to characterize hypoxic events in Gulf of Mexico estuaries. They found that single, daytime, instantaneous measures of DO detected hypoxia only 20% of the time that it was known to occur based on 31 days of continuous (15-minute intervals) sampling data. When randomly selected 24-96 hour periods of the continuous data were used, hypoxia was indicated 50% of the time it was detected in the full data set. A similar comparison was performed using Tampa Bay data (Janicki et al. 2001). Tampa Bay monitoring programs typically measure DO between 9 am and 2 pm. When mid-day measurements were compared to DO values collected over full 24 h periods, 25% of the mid-day observations failed to detect hypoxia that occurred during some other portion of the day. While Tampa Bay mid-day DO measurements reflected hypoxic events more accurately than the data used by Summers and Engle (1993), daytime DO measurements clearly cannot detect hypoxic events a significant portion of the time. Hypoxia can be so variable both within and between days that long-term continuous monitoring is needed to detect hypoxia (Summers and Engle 1993). While Summers and Engle (1993) used 15-minute interval data, IRWD's 1997-2000 study in UNB seemed to adequately capture hypoxic events with 30-minute intervals. The frequency of sampling needed to characterize hypoxic events in Newport Bay should be investigated so that the system can be adequately characterized.

Measurement of DO at various depths is also critical in identifying hypoxia. Data from OCPFRD show vertical gradients in DO in UNB with surface water DO being above suggested thresholds while bottom waters are depleted. Data from the Chesapeake Bay (Kemp and Boynton 1980) also demonstrated this pattern, with DO concentrations dropping from 10.5 mg/l in surface waters to 2.3 mg/l in bottom waters.

Consideration of sensitive habitat may be another factor in designing a DO monitoring plan. Hypoxia may not occur frequently in Newport Bay relative to other systems, yet the areas in which it does occur might be critical habitat for important species in the Bay. The significance of hypoxia in Newport Bay may vary with habitat.

#### Recommendations for Dissolved Oxygen Criteria Development

To begin development of defensible DO criteria with which to manage water quality in Newport Bay, the following considerations need to be explored:

- The links between water quality and DO need to be substantiated if DO criteria are to be used as a metric of water quality. The relationships between nutrient loading, macroalgae and DO need to be investigated, characterized and quantified if possible.
- The effects of DO on Newport Bay species need to be established so that DO criteria protective of aquatic life can be developed.
- A better understanding of how the predominance of macroalgae (as opposed to phytoplankton) in Newport Bay makes the system different from other systems where water quality and overall ecosystem health is needed.

- The role and importance of vertical stratification in relation to hypoxia in Newport Bay needs to be understood.
- The natural variability in DO needs to be characterized.
- A monitoring plan should be devised that maximizes the chance of capturing the sensitive areas, periods, and durations when hypoxia occurs.

In conclusion, it is important to remember that low DO availability often occurs in conjunction with other anthropogenic stressors that accompany increased development. These additional factors, which include increased pathogen prevalence, fishing pressure, sediment loads, increased and altered freshwater inputs, and hydrodynamic modifications (Breitburg 2002), may interact synergistically with hypoxia to negatively impact aquatic resources. As DO is considered to be indicative of a problem rather than a problem that can be regulated directly, a watershed management plan addressing the activities and processes in the watershed that contribute to hypoxia may result in reductions in these other problems as well.

Table 6.1. Response of adult Chesapeake Bay macrobenthic species to low dissolved oxygen concentrations. Chesapeake Bay species listed are in same genera as organisms in Upper Newport Bay, CA. UNB species listed in CDFG (1989); Chesapeake Bay species data from multiple sources cited in US EPA (2003).

UNB Species	Chesapeake Bay Species	DO (mg/l)	Temperature (°C)	Observed Response
<b>Mollusca</b>				
<i>Macoma carolottensis</i> , <i>M. identata</i> , <i>M. inquinata</i> , <i>M. nasuta</i>	<i>Macoma balthica</i>	0	10	4% mortality in 7 days
		0	10	50% mortality in 21 days
<i>Mytilus edulis</i>	<i>Mytilus edulis</i>	0.2	10	50% mortality in 35 days without sulfide, in 25 days with sulfide
		0	10	20% mortality in 7 days
<b>Polychaeta</b>				
<i>Capitella capitata</i>	<i>Capitella capitata</i>	0	12	Mortality in 8 days
<i>Eteone alba</i> , <i>E. dilatae</i>	<i>Eteone picta</i>	0	12	Mortality in 6 days
<i>Glycera americana</i> , <i>G. oxycephala</i>	<i>Glycera convoluta</i>	0	12	Mortality in 10 days
<i>Nephtys caecoides</i> , <i>N. californiensis</i> , <i>N. cornuta franciscana</i>	<i>Nephtys ciliata</i>	0	10	50% mortality in 6 days
<i>Nereis acuta</i> , <i>N. dendritica</i> , <i>N. eakini</i> , <i>N. procera</i>	<i>Nereis diversicolor</i>	0.2-0	10	50% mortality in 5 days without sulfide, in 4 days with sulfide
		0	6-8	No mortality but ATP concentration 59% of initial after 72 hours
	<i>Nereis pelagica</i>	0	6-8	40% mortality after 36 hours; ATP concentration 51% of initial after 72 hours
	<i>Nereis virens</i>	0	6-8	No mortality but ATP concentration 57% of initial after 72 hours
<b>Family Terebellidae</b>	<i>Terebellides stroemi</i>	0	10	50% mortality in 3 days

Table 6.2. Results of Chesapeake Bay fish and invertebrate low dissolved oxygen acute sensitivity tests. Chesapeake Bay species listed are in same genera as organisms in Upper Newport Bay, CA. UNB species listed in CDFG (1989); Chesapeake Bay species data from multiple sources cited in US EPA (2000).

<i>UNB Species</i>	<i>Chesapeake Bay Species</i>	<i>Parameter</i>	<i>Dissolved Oxygen Concentration (mg/l)</i>
<b>California halibut</b> <i>Paralichthys californicus</i>	<b>Summer flounder</b> <i>Paralichthys dentatus</i>	24 h LC50 <sup>a</sup>	1.10
		96 h LC50	1.10
		SMAV <sup>b</sup> LC50	1.32
		SMAV LC5	1.57
<b>Pipefish (kelp, bay, and barred)</b> <i>Syngnathus californiensis</i> , <i>S. leptorhynchus</i> , <i>S. auliscus</i>	<b>Pipefish</b> <i>Syngnathus fuscus</i>	SMAV LC50	1.63
		SMAV LC5	1.9
<b>California killifish</b> <i>Fundulus parvipinnis</i>	<b>Mummichog</b> <i>Fundulus heteroclitus</i>	10% embryo mortality in 1 day	4.5
<b>Polychaete</b> <i>Nereis acuta</i> , <i>N. dendritica</i> , <i>N. eakini</i> , <i>N. procera</i>	<b>Polychaete</b> <i>Nereis diversicolor</i>	Feeding stopped	1.2
	<i>Nereis virens</i>	Emergence from sediment	0.9
		Feeding stopped	0.9
<b>Amphipod</b> <i>Ampelisca cristata</i>	<b>Amphipod</b> <i>Ampelisca abdita</i>	SMAV LC50	<0.9

<sup>a</sup>LC50 and LC5 are the lethal concentrations of oxygen for 50 and 5% of the test population, respectively.

<sup>b</sup>Species Mean Acute Value

Table 6.3. Non-species specific dissolved oxygen criteria and responses of groups of organisms to low dissolved oxygen concentrations. From US EPA (2003).

<i>Protected process/ Organism response</i>	<i>Dissolved oxygen (mg/l)</i>	<i>Time period</i>
<b>Egg/larval recruitment</b>	3 1.7	30 days Instantaneous minimum
<b>Juvenile/adult survival</b>	>2.3	24 hours
<b>Zooplankton habitat avoidance Reduced copepod nauplii abundance</b>	<1	-
<b>Reduced copepod survival</b>	<0.86-1.3	24 hours
<b>100% copepod mortality</b>	0.71	24 hours

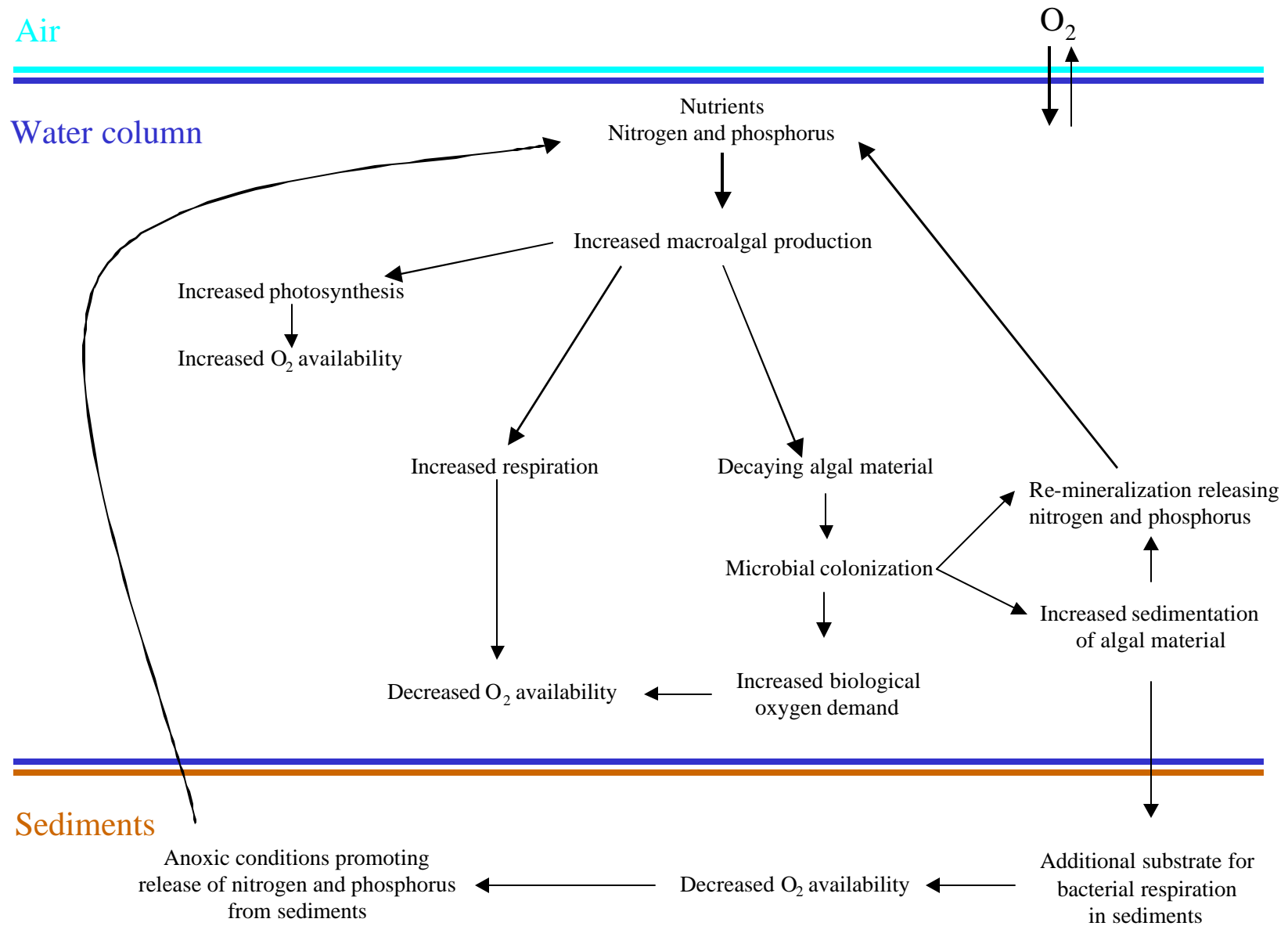


Figure 6.1

LITERATURE CITED

- Allen, L. G. 1994. The effect of the deep water and sediment control project, Unit III, on the fishes of Upper Newport Bay. California State University. Northridge, CA.
- Alex Horne Associates (AHA). 1998. Effect of macroalgae (seaweeds) on impairment of beneficial uses of Newport Bay and quantification of nutrient control levels needed in San Diego Creek to remove beneficial use impairment. Prepared for Irvine Ranch Water District by Alex Horne Associates. El Cerrito, CA.
- Aneer, G. 1985. Some speculations about the Baltic herring (*Clupea harengus membras*) in connection with the eutrophication of the Baltic Sea. Canadian Journal of Fisheries and Aquatic Science 42: 83-90.
- American Public Health Association, American Water Works Association, and Water Environmental Federation (APHA). 1998. Oxygen (dissolved), p. 4 - 129 - 4-136. In L. S. Clesceri, A. E. Greenberg and A. D. Eaton [eds.], Standard methods for the examination of water and wastewater, 20th edition.
- Bolam, S. G., T. F. Fernandes, P. Read, and D. Raffaelli. 2000. Effects of macroalgal mats on intertidal sandflats: an experimental study. Journal of Experimental Marine Biology and Ecology 249: 123-137.
- Boyer, K. E. 2002. Linking community assemblages and ecosystem processes in temperate and tropical coastal habitats. Ph.D. Thesis, Organismic Biology, Ecology and Evolution. University of California, Los Angeles.
- Breitburg, D. L. 1994. Behavioral response of fish larvae to low dissolved oxygen concentrations in a stratified water column. Marine Biology 120(4): 615-625.
- Breitburg, D. L. 2002. Effects of hypoxia, and the balance between hypoxia and enrichment, on coastal fishes and fisheries. Estuaries 25(4b): 767-781.
- California Department of Fish and Game (CDFG). 1989. Management plan Upper Newport Bay Ecological Reserve.
- Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210: 223-253.
- D'Avanzo, C., and J. N. Kremer. 1994. Diel oxygen dynamics and anoxic events in an eutrophic estuary of Waquoit Bay, Massachusetts. Estuaries 17(1B): 131-139.
- Desmond, J., M. Cordrey, J. Johnson, K. Ward, and J. B. Zedler. 2000. Tijuana River National Estuarine Research Reserve: annual report on ecosystem monitoring. Annual report for the year ending 31 December, 1999. Pacific Estuarine Research Laboratory, San Diego State University, San Diego, CA.
- Desmond, J., G. Williams, M. James, J. Johnson, J. Callaway, and J. B. Zedler. 1999. Tijuana River National Estuarine Research Reserve: annual report on ecosystem monitoring. Annual report for the year ending 31 December, 1998. Pacific Estuarine Research Laboratory, San Diego State University, San Diego, CA.
- Diaz, R. J., and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. Oceanography and Marine Biology: an Annual Review 33: 245-303.
- Flindt, M. R., M. A. Pardal, A. I. Lillebo, I. Martins, and J. C. Marques. 1999. Nutrient cycling and plant dynamics in estuaries: A brief review. Acta Oecologica 20(4): 237-248.

- Fong, P., R. M. Donohoe, and J. B. Zedler. 1994. Nutrient concentration in tissue of the macroalga *Enteromorpha* as a function of nutrient history: An experimental evaluation using field microcosms. *Marine Ecology Progress Series* 106(3): 273-281.
- Fong, P., J. B. Zedler, and R. M. Donohoe. 1993. Nitrogen vs. phosphorus limitation of algal biomass in shallow coastal lagoons. *Limnology and Oceanography* 38(5): 906-923.
- Fujita, R. M. 1985. The role of nitrogen status in regulating transient ammonium uptake and nitrogen storage by macroalgae. *Journal of Experimental Marine Biology and Ecology* 92: 283-301.
- Hanisak, M. D. 1983. The nitrogen relationships of marine macroalgae, p. 699-730. *In* E. J. Carpenter and D. C. Capone [eds.], *Nitrogen in the marine environment*. Academic Press.
- Harlin, M. M., and B. Thorne-Miller. 1981. Nutrient enrichment of seagrass beds in a Rhode Island coastal lagoon. *Marine Biology* 65: 221-229.
- Hauxwell, J., J. McClelland, P. J. Behr, and I. Valiela. 1998. Relative importance of grazing and nutrient controls of macroalgal biomass in three temperature shallow estuaries. *Estuaries* 21(2): 347-360.
- Hernández, I., G. Peralta, J. L. Perez-Llorens, J. J. Vergara, and F. X. Niell. 1997. Biomass and dynamics of growth of *Ulva* species in Palmones River estuary. *Journal of Phycology* 33(5): 764-772.
- Howarth, R. W. 1988. Nutrient limitation of net primary production in marine ecosystems. *Annual Review of Ecology and Systematics* 19: 89-110.
- Hull, S. C. 1987. Macroalgal mats and species abundance: a field experiment. *Estuarine Coastal and Shelf Science* 25: 519-532.
- Janicki, A., R. Pribble, and M. Winowitch. 2001. Examination of the spatial and temporal nature of hypoxia in Tampa Bay, Florida. Prepared for Tampa Bay Estuary Program by Janicki Environmental, Inc. St. Petersburg, FL. Tampa Bay Estuary Program Technical Publication #09-01.
- Jorgensen, B. B. 1996. Material flux in the sediment, p. 115-135 *In* B. B. Jorgensen and K. Richardson [eds.], *Eutrophication in coastal marine ecosystems*. Coastal and Estuarine Studies. American Geophysical Union.
- Kamer, K., K. A. Boyle, and P. Fong. 2001. Macroalgal bloom dynamics in a highly eutrophic southern California estuary. *Estuaries* 24(4): 623-635.
- Kemp, W. M., and W. M. Boynton. 1980. Influence of biological and physical processes on dissolved oxygen dynamics in an estuarine system: implications for measurement of community metabolism. *Estuarine and Coastal Marine Science* 11: 407-431.
- Krause-Jensen, D., P. B. Christensen, and S. Rysgaard. 1999. Oxygen and nutrient dynamics within mats of the filamentous macroalga *Chaetomorpha linum*. *Estuaries* 22(1): 31-38.
- Kwak, T. J., and J. B. Zedler. 1997. Food web analysis of southern California coastal wetlands using multiple stable isotopes. *Oecologia (Berlin)* 110(2): 262-277.
- Lavery, P. S., and A. J. McComb. 1991. Macroalgal-sediment nutrient interactions and their importance to macroalgal nutrition in a eutrophic estuary. *Estuarine Coastal and Shelf Science* 32: 281-295.
- Marcomini, A., A. Sfriso, B. Pavoni, and A. A. Orio. 1995. Eutrophication of the Lagoon of Venice: nutrient loads and exchanges, p. 59-79. *In* A. J. McComb [ed.], *Eutrophic shallow estuaries and lagoons*. CRC Press.

- McComb, A. J., R. P. Atkins, P. B. Birch, D. M. Gordon, and R. J. Lukatelich. 1981. Eutrophication in the Peel-Harvey estuarine system, western Australia, p. 323-342. In B. J. Neilson and L. E. Cronin [eds.], *Estuaries and Nutrients*. Humana Press.
- Mitsch, W. J., and J. G. Gosselink. 1993. *Wetlands*. Van Nostrand Reinhold, New York.
- Nixon, S. W. 1988. Physical energy inputs and the comparative ecology of lake and marine ecosystems. *Limnology and Oceanography* 33(4, part 2): 1005-1025.
- Nixon, S. W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* 41(0): 199-219.
- National Oceanic and Atmospheric Administration (NOAA). 1998 (on-line). Oxygen depletion in coastal waters. NOAA. Silver Spring, MD.
- Norkko, A., and E. Bonsdorff. 1996. Rapid zoobenthic community responses to accumulations of drifting algae. *Marine Ecology Progress Series* 131: 143-157.
- Nowicki, B. L., and S. W. Nixon. 1985. Benthic nutrient remineralization in a coastal lagoon ecosystem. *Estuaries* 8(2B): 182-190.
- Orange County Human Services Agency, Public Health and Medical Services, Environmental Health Division, and Water Quality Control Section (OCHSA). 1978. Environmental studies in Newport Bay. Environmental Health Division.
- Parker, C. A., and J. E. O'Reilly. 1991. Oxygen depletion in Long Island Sound: a historical perspective. *Estuaries* 14(3): 248-264.
- Peckol, P., B. DeMeo-Anderson, J. Rivers, I. Valiela, M. Maldonado, and J. Yates. 1994. Growth, nutrient uptake capacities and tissue constituents of the macroalgae *Cladophora vagabunda* and *Gracilaria tikvahiae* related to site-specific nitrogen loading rates. *Marine Biology (Berlin)* 121(1): 175-185.
- Pedersen, M. F. 1994. Transient ammonium uptake in the macroalga *Ulva lactuca* (Chlorophyta): nature, regulation, and the consequences for choice of measuring technique. *Journal of Phycology* 30: 980-986.
- Public Facilities & Resources Department Orange County California (PFRD). 2001. Report of the Regional Monitoring Program for the Newport/San Diego Creek watershed nutrient TMDL. County of Orange and Cities of Irvine, Tustin, Newport Beach, Lake Forest, Santa Ana, Orange and Costa Mesa.
- Rabalais, N. N., R. E. Turner, and W. J. J. Wiseman. 2002. Gulf of Mexico hypoxia, a.k.a. "The dead zone". *Annual Review of Ecology and Systematics* 33: 235-263.
- Rabalais, N. N., W. J. J. Wiseman, and R. E. Turner. 1994. Comparison of continuous records of near-bottom dissolved oxygen from the hypoxia zone along the Louisiana coast. *Estuaries* 17(4): 850-861.
- Raffaelli, D., S. Hull, and H. Milne. 1989. Long-term changes in nutrients, weed mats and shorebirds in an estuarine system. *Cahiers de Biologie Marine* 30(2): 259-270.
- Raffaelli, D., J. Limia, S. Hull, and S. Pont. 1991. Interactions between the amphipod *Corophium volutator* and macroalgal mats on estuarine mudflats. *Journal of the Marine Biological Association of the United Kingdom* 71(4): 899-908.
- Resource Management Associates Inc. (RMA). 2001. Newport Bay Water Quality Model Development 3-D Stratified Flow Analysis. Prepared for California State Water Resources Control Board. Suisun City, CA.
- Rosenberg, G., and J. Ramus. 1984. Uptake of inorganic nitrogen and seaweed surface area:volume ratios. *Aquatic Botany* 19: 65-72.

- Schramm, W., and P. H. Nienhuis, (Eds.). 1996. Marine benthic vegetation: recent changes and the effects of eutrophication. Springer-Verlag, New York. 470 pp.
- Sfriso, A., A. Marcomini, and B. Pavoni. 1987. Relationships between macroalgal biomass and nutrient concentrations in a hypertrophic area of the Venice Lagoon Italy. *Marine Environmental Research* 22(4): 297-312.
- Sfriso, A., B. Pavoni, A. Marcomini, and A. A. Orio. 1992. Macroalgae, nutrient cycles, and pollutants in the Lagoon of Venice. *Estuaries* 15(4): 517-528.
- Smith, K. N., and W. F. Herrnkind. 1992. Predation on early juvenile spiny lobsters *Panulirus argus* (Latreille): influence of size and shelter. *Journal of Experimental Marine Biology and Ecology* 157: 3-18.
- Soulsby, P. G., D. Lowthion, and M. Houston. 1982. Effects of macroalgal mats on the ecology of intertidal mudflats. *Marine Pollution Bulletin* 13(5): 162-166.
- Stanley, D. W., and S. W. Nixon. 1992. Stratification and bottom-water hypoxia in the Pamlico River Estuary. *Estuaries* 15(3): 270-281.
- Stephan, C. E., D. I. Mount, D. J. Hansen, G. H. Gentile, G. A. Chapman, and W. A. Brungs. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. NTIS Publication No.: PB85-227049.
- Summers, J. K., and V. D. Engle. 1993. Evaluation of sampling strategies to characterize dissolved oxygen conditions in northern Gulf of Mexico estuaries. *Environmental Monitoring and Assessment* 24: 219-229.
- Tenore, K. R. 1977. Utilization of aged detritus derived from different sources by the polychaete *Capitella capitata*. *Marine Biology* 44: 51-55.
- Thrush, S. F. 1986. The sublittoral macrobenthic community structure of an Irish sea-lough: effect of decomposing accumulations of seaweed. *Journal of Experimental Marine Biology and Ecology* 96: 199-212.
- Turner, R. E., W. W. Schroeder, and W. J. Wiseman, Jr. 1987. The role of stratification in the deoxygenation of Mobile Bay and adjacent shelf bottom waters. *Estuaries* 10(1): 13-19.
- U.S. Environmental Protection Agency (US EPA). 1986. Ambient water quality criteria for dissolved oxygen. Office of Water. 440-586-003.
- U.S. Environmental Protection Agency (US EPA). 1998. Total maximum daily loads for nutrients San Diego Creek and Newport Bay, California. US EPA Region 9.
- U.S. Environmental Protection Agency (US EPA). 2000. Ambient aquatic life water quality criteria for dissolved oxygen (saltwater): Cape Cod to Cape Hatteras. EPA-822-R-00-012. Office of Water, Office of Science and Technology, Washington, DC, and Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett, RI.
- U.S. Environmental Protection Agency (US EPA). 2001. Nutrient criteria technical guidance manual: estuarine and coastal marine waters. EPA-822-B-01-003. Office of Water, Office of Science and Technology, Washington, D.C.
- U.S. Environmental Protection Agency (US EPA). 2003. Ambient water quality criteria for dissolved oxygen, water clarity and chlorophyll a for the Chesapeake Bay and its tidal tributaries. EPA 903-R-03-002. US EPA Region III, Annapolis, MD, Region III Water Protection Division, Philadelphia, PA, and Office of Water, Office of Science and Technology, Washington, D.C.
- Valiela, I., K. Foreman, M. LaMontagne, D. Hersh, J. Costa, P. Peckol, B. DeMeo-Andreson, C. D'Avanzo, M. Babione, C. H. Sham, J. Brawley, and K. Lajtha. 1992. Couplings of

- watersheds and coastal waters sources and consequences of nutrient enrichment in Waquoit Bay Massachusetts. *Estuaries* 15(4): 443-457.
- Valiela, I., J. McClelland, J. Hauxwell, P. J. Behr, D. Hersh, and K. Foreman. 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography* 42(5 PART 2): 1105-1118.
- Wannamaker, C. M., and J. A. Rice. 2000. Effects of hypoxia on movements and behavior of selected estuarine organisms from the southeastern United States. *Journal of Experimental Marine Biology and Ecology* 249: 145-163.
- Ward, K. M., M. Cordrey, and J. M. West. 2000. The physical, chemical and biological monitoring of Los Penasquitos Lagoon 21 September 1999-20 September 2000. Pacific Estuarine Research Laboratory, San Diego State University. San Diego, CA.
- Ward, K. M., J. M. West, and M. Cordrey. 2001. The physical, chemical and biological monitoring of Los Penasquitos Lagoon 21 September 2000-20 September 2001. Pacific Estuarine Research Laboratory, San Diego State University. San Diego, CA.
- Welsh, B. L., and F. C. Eller. 1991. Mechanisms controlling summertime oxygen depletion in western Long Island Sound. *Estuaries* 14(3): 265-278.
- West, J. M., M. Cordrey, S. P. Madon, and J. B. Zedler. 2001. Tijuana River National Estuarine Research Reserve: report on ecosystem monitoring. Report of monitoring conducted January 1-June 30, 2001. Pacific Estuarine Research Laboratory, San Diego State University. San Diego, CA.
- West, J. M., M. Cordrey, S. P. Madon, and J. B. Zedler. 2002. Tijuana River National Estuarine Research Reserve: annual report on ecosystem monitoring. Annual report of monitoring conducted July 1, 2001-June 30, 2002. Pacific Estuarine Research Laboratory, San Diego State University, San Diego, CA.
- Williams, G. D., Principal Author. 1997. The physical, chemical and biological monitoring of Los Penasquitos Lagoon 20 September 1996-20 September 1997. Pacific Estuarine Research Laboratory, San Diego State University. San Diego, CA.
- Williams, G., (principal author), J. Callaway, K. Ward, J. Johnson, M. Cordrey, and J. B. Zedler. 1998. Tijuana River National Estuarine Research Reserve: annual report on ecosystem monitoring. Annual report for the year ending 31 December, 1997. Pacific Estuarine Research Laboratory, San Diego State University, San Diego, CA.
- Williams, G. D., J. M. West, M. Cordrey, and K. Ward. 1999. The physical, chemical and biological monitoring of Los Penasquitos Lagoon 20 September 1998-20 September 1999. Pacific Estuarine Research Laboratory, San Diego State University. San Diego, CA.
- Wilson, K. A., K. W. Able, and K. L. J. Heck. 1990. Predation rates on juvenile blue crabs in estuarine nursery habitats: evidence for the importance of macroalgae (*Ulva lactuca*). *Marine Ecology Progress Series* 58: 243-251.
- Zedler, J. B. 1980. Algal mat productivity: comparisons in a salt marsh. *Estuaries* 3(2): 122-131.
- Zimmerman, C. F., and J. R. Montgomery. 1984. Effects of a decomposing drift algal mat on sediment pore water nutrient concentrations in a Florida seagrass bed. *Marine Ecology Progress Series* 19: 299-302.

APPENDIX A  
MAPS

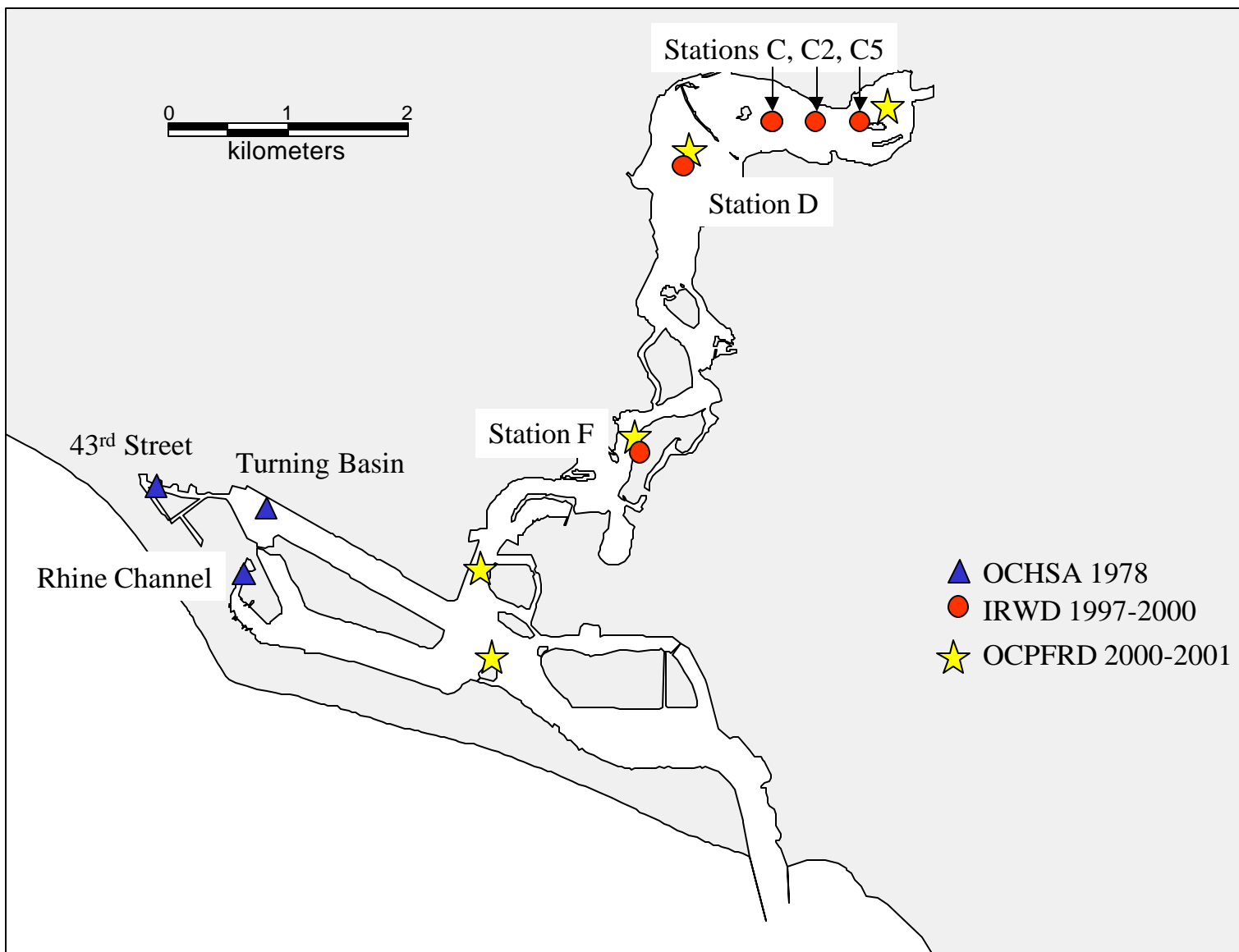


Figure A1. Map of Newport Bay stations in 3 different studies in which dissolved oxygen was measured.



Figure A2. Map of 5 estuaries sampled by UCLA and SCCWRP.

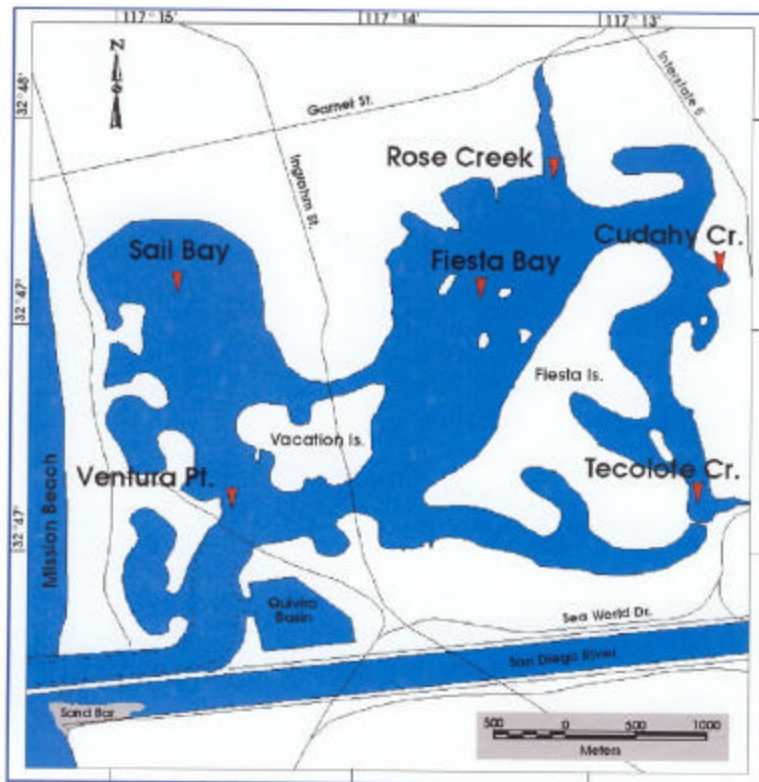


Figure A3. Map of sampling stations in Mission Bay, courtesy of R. Kaufmann, University of San Diego.